

The University of San Francisco

USF Scholarship: a digital repository @ Gleeson Library | Geschke Center

Master's Projects and Capstones

Theses, Dissertations, Capstones and Projects

Spring 5-15-2020

Impacts of medusahead (*Elymus caput-medusae* L.) management on plant communities in California's valley grasslands

Nicole Carpenter
nmcarpenter@usfca.edu

Follow this and additional works at: <https://repository.usfca.edu/capstone>



Part of the [Natural Resources and Conservation Commons](#)

Recommended Citation

Carpenter, Nicole, "Impacts of medusahead (*Elymus caput-medusae* L.) management on plant communities in California's valley grasslands" (2020). *Master's Projects and Capstones*. 1034.
<https://repository.usfca.edu/capstone/1034>

This Project/Capstone is brought to you for free and open access by the Theses, Dissertations, Capstones and Projects at USF Scholarship: a digital repository @ Gleeson Library | Geschke Center. It has been accepted for inclusion in Master's Projects and Capstones by an authorized administrator of USF Scholarship: a digital repository @ Gleeson Library | Geschke Center. For more information, please contact repository@usfca.edu.

This Master's Project

Impacts of medusahead (*Elymus caput-medusae* L.) management on plant communities in California's valley grasslands

by

Nicole Carpenter

is submitted in partial fulfillment of the requirements
for the degree of:

**Master of Science
in
Environmental Management**

at the

University of San Francisco

Submitted:

Received:

.....
Nicole Carpenter Date

.....
Aviva Rossi, M.S. Date

Table of Contents

List of Figures.....	ii
List of Tables	iii
Abstract.....	iv
1. Introduction.....	1
<i>1.1 Objective of this Study</i>	<i>3</i>
2. Background	4
<i>2.1 California's Valley Grasslands</i>	<i>4</i>
<i>2.2 Ecology of Medusahead.....</i>	<i>7</i>
<i>2.3 Impacts of Medusahead.....</i>	<i>9</i>
2.3.1 Loss of Biodiversity	9
2.3.2 Forage Decline	10
2.3.3 Thatch Accumulation	10
2.3.4 Economic Cost	11
<i>2.4 Medusahead Control Methods.....</i>	<i>11</i>
3. Methods.....	12
4. Results	13
<i>4.1 Herbicides.....</i>	<i>18</i>
4.1.1 Advantages	19
4.1.2 Disadvantages.....	20
4.1.3 Impacts on Plant Community	21
<i>4.2 Grazing.....</i>	<i>22</i>
4.2.1 Advantages	24
4.2.2 Disadvantages.....	25
4.2.3 Impacts on Plant Community	26
<i>4.3 Prescribed Burning</i>	<i>28</i>
4.3.1 Advantages	28
4.3.2 Disadvantages.....	29
4.3.3 Impacts on Plant Community	30
<i>4.4 Mechanical control.....</i>	<i>33</i>
4.4.1 Advantages	34
4.4.2 Disadvantages.....	34
4.4.1 Impacts on Plant Community	35
5. Discussion.....	36

5.1 Integrated Pest Management.....	39
5.2 The Knowing-Doing Gap	40
5.3 Revegetation Implementation	42
5.4 Climate Change and Drought.....	43
5.5 Study Limitations	44
6. Conclusion and Management Recommendations	45
6.1 Restoration Research.....	46
6.2 Funding for Weed Management Areas	46
6.3 Monitoring and Adaptive Management.....	47
References.....	49

List of Figures

Figure 1: The invasion curve	2
Figure 2: Current distribution of California's grasslands	5
Figure 3: Medusahead seed head with visible awns	8
Figure 4: Medusahead distribution in California.....	9
Figure 5: Medusahead thatch	11
Figure 6: Medusahead control with the pre-emergent application of the herbicides.....	20
Figure 7: The Green and Brown grazing strategy.....	23
Figure 8: Spring timing of sheep grazing on medusahead cover.....	24
Figure 9: Medusahead cover from grazed and ungrazed plots	26
Figure 10: Effect of grazing densities and timing on response ratios of medusahead, native plants, and non-native forage plants	27
Figure 11: Percent cover of thatch and bare ground between unburned and burned plots in a vernal pool ecosystem.....	29
Figure 12: Species abundance and dominant species at two study sites two years post-burn.....	31
Figure 13: Percent cover of burned and unburned plots.	32
Figure 14: Conceptual framework for integrated pest management in a grassland ecosystem	40
Figure 15: Usability of scientific literature by land managers.....	41

List of Tables

Table 1: Synthesis table	14
Table 2: Characteristics of herbicides utilized in medusahead control studies	19
Table 3: Transitions between dominant vegetation types from pre-treatment to post-treatment..	36
Table 4: Overview of medusahead management goals and appropriate control strategies	37

Abstract

California's valley grasslands are one of the most invaded ecosystems in the state. It is estimated that valley grasslands contain between 90 to 99% cover of non-native plants. The most recent wave of invasive plants has included medusahead (*Elymus caput-medusae* L.). Medusahead is an annual grass that matures two to four weeks later than most other grasses. Management of medusahead includes the use of herbicides, targeted grazing, prescribed burns, and mechanical control. The primary focus of most studies on the use of these management methods is on the control of medusahead rather than the impacts on non-target plants. This study examines published research to determine what impact medusahead management has on the composition of plant communities within valley grasslands. Herbicides have mixed impacts on the percent cover of grasses and forbs. Targeted grazing resulted in decreased percent cover of non-native grasses and increased forb cover. Grazing results in neutral to increased percent cover of native plants. Prescribed burns decreased the percent cover of non-native grasses, increased forb cover, and had mixed impacts on native plant cover. Mechanical control shifted vegetative states towards forb or filaree (*Erodium* spp.) dominated communities. The non-target impacts of medusahead control were generally short-lived with differences in percent cover returning to baseline conditions within one to three years. To mitigate the non-target impacts of medusahead, revegetation efforts should be prioritized in sites with higher abundances of native plant species. The future success of controlling medusahead is dependent upon grassland restoration research, consistent funding for weed management areas to aid in managing invasive plants, and the implementation of monitoring after medusahead control treatments.

1. Introduction

Invasive plants are species that do not originate from a region they now inhabit and can quickly spread throughout an environment (Clinton 1999). These plants may cause economic and environmental damage or threaten human health (Clinton 1999). Invasive species are the second greatest threat to biodiversity behind habitat loss (Wilcove et al. 1998). In an ecological context, invasive plants often outcompete and displace native plants and change the structure of a plant community (Mack et al. 2000). These alterations can decrease native wildlife diversity (Fletcher et al. 2019), alter ecosystem nutrient cycling (Liao et al. 2008), and reduce rangeland forage (DiTomaso 2000). On an annual basis, invasive plants cause \$1 billion worth of losses and damages and cost \$5 billion in management to rangelands in the United States (U.S.) (Pimentel et al. 2005). Therefore, managing invasive plants is a high priority for many land managers (Schohr et al. 2020).

The management strategies employed by land managers to control invasive plants is based on an analysis of the threat of an invasion on their land versus the limited resources available to deal with such invasions (Sheley and Smith 2012). The most common management strategies are preventing the establishment of an invasive, early detection and eradication of established populations, containment of newly established populations, and the long-term management of ecosystems dominated by an invasive plant (Harvey and Mazzotti 2014). The economic cost of controlling invasive plants increases as the severity of invasion increases (Figure 1) (Harvey and Mazzotti 2014). Generally, preventing the establishment of an invasive plant in new areas is the most cost-effective management strategy (DiTomaso 2000). When an invasive plant species becomes widespread it can procure a higher cost to contain the infestation and prevent the invasion from expanding even further (Sheley and Smith 2012).

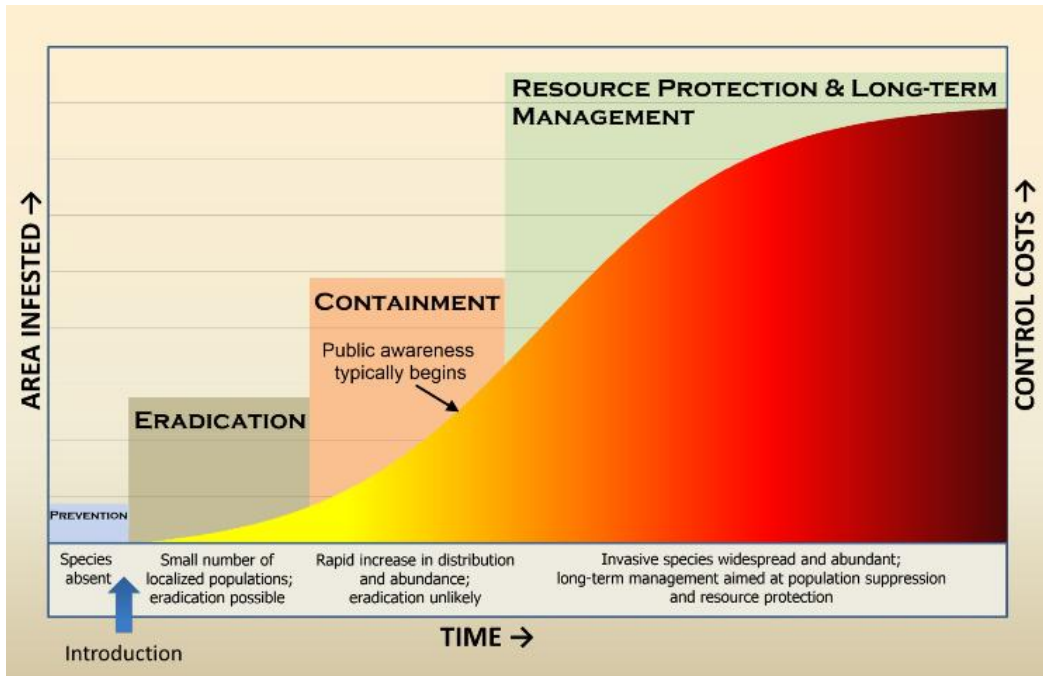


Figure 1: The invasion curve (Harvey and Mazzotti 2014)

Invasive plants often have a detrimental impact on native plants and animals within an invaded ecosystem (Rinella et al. 2009). However, management actions taken to reduce invasive plants may also have unintended consequences for native species, such as a decrease in population size (Rinella et al. 2009). For example, the endangered Ridgway's Rail (*Rallus obsoletus obsoletus*) has become reliant upon dense stands of invasive cordgrass (*Spartina foliosa* x *alterniflora*) for nesting habitat due to the displacement of native cordgrass (*Spartina alterniflora*) by the invasive cordgrass (Overton et al. 2014). Restoring dense stands of native cordgrass after the invasive cordgrass was removed has proven to be an extremely slow process (Lampert et al. 2014). A management strategy that would completely eradicate the invasive cordgrass prior to restoration of sufficient areas with the native cordgrass would be detrimental to the Ridgway's Rail (Lampert et al. 2014). The optimal management strategy to control the invasive cordgrass while minimizing the detrimental impacts on the Ridgway's Rail would be to intersperse the establishment of native cordgrass populations with the removal of invasive populations of cordgrass (Lampert et al. 2014). While an optimal management strategy is available for mitigating this invasive cordgrass, in other ecosystems the positive and negative impacts of invasive plants and the management strategies used to control them are largely unknown (Skurski et al. 2013).

California's grasslands are considered the most invaded ecosystem within the state (Bossard and Randall 2007). Most grasslands in California contain between 90 to 99% cover of non-native plants (Bartolome et al. 2007). These grasslands are estimated to contain approximately 37% of the state's invasive plant species (Bossard and Randall 2007). The biological invasion of California's grasslands began in the 1700s with the arrival of Spanish settlers and continued into the 1800s with the arrival of Mexican and European ranchers (Burcham 1961). The Gold Rush of 1849 brought a dramatic increase in the population size of California and resulted in an increase of cattle ranches on California's grasslands (Burcham 1961). In addition to cattle, the ranchers brought European grass species for forage with them (Burcham 1961). As a result, a surge of non-native annual grasses established over a twenty to thirty-year span in the mid-1800s and eventually led to a landscape dominated by non-native plants (Burcham 1961). It is thought that native perennial grasses were weakened and outcompeted due to persistent drought and intensive, year-round grazing (Bartolome et al. 2007). Introduced, annual grasses, such as medusahead (*Elymus caput-medusae*), continue to be a dominant species in most of California's grasslands (Bartolome et al. 2007). These invaded grasslands often require management to lessen the negative impacts caused by these species (DiTomaso and Smith 2012).

1.1 Objective of this Study

This study assesses the impacts of managing the invasive plant, medusahead, on the composition of plant communities within California's valley grasslands. This study compares the following management strategies: herbicides, grazing, prescribed burning, and mechanical control. These management strategies will be evaluated through a comparative analysis of the current literature. I will discuss how medusahead management impacts valley grassland plant communities. I also provide recommended management strategies for medusahead and topics for future research.

2. Background

2.1 California's Valley Grasslands

The most widespread of California's grasslands are the valley grasslands (Eviner 2016). These grasslands are also referred to as interior or annual grasslands (Eviner 2016). Valley grasslands are defined by the presence of herbaceous grasses and forbs (i.e., non-grasses) and a lack of woody species, such as trees or shrubs (Bartolome et al. 2007). Historically, valley grasslands covered most of the Central Valley of California before they were converted to agricultural fields (Bartolome et al. 2007). The Central Valley is a large lowland area that dominates the middle of the state and is bordered by the Coast Range on the west and the Sierra Nevada on the east. It is thought that these historic grasslands were dominated by native perennial bunchgrasses that can live upwards of 100 years (Bartolome et al. 2007, Hamilton et al. 2002). Currently, the remaining valley grasslands form a belt surrounding the agricultural regions of the Central Valley and span from Redding in the north to Bakersfield in the south (Figure 2) (Keeler-Wolf et al. 2007). Valley grasslands can extend up to 700 m in elevation into the foothills of the Sierra Nevada and Coastal Ranges and to the southern coastal areas near Santa Barbara (Keeler-Wolf et al. 2007).

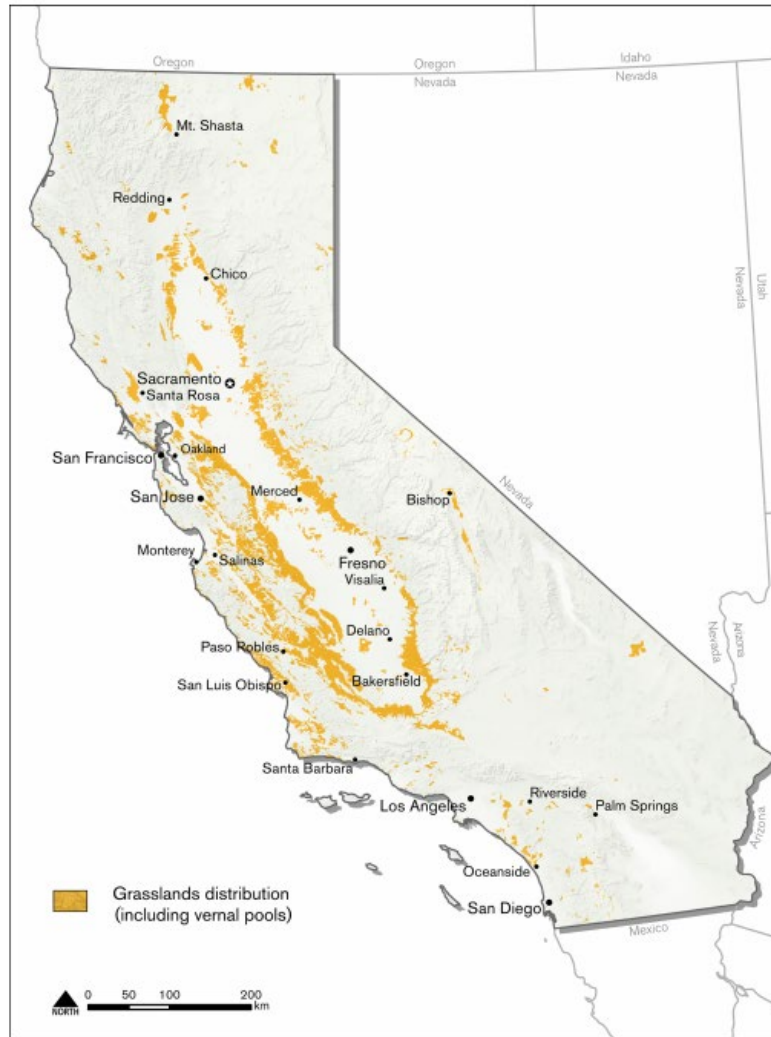


Figure 2: Current distribution of California’s grasslands. Grasses within oak woodland and savannah ecosystems are not depicted. Valley grasslands primarily occur in the area surrounding the Central Valley (Eviner 2016).

Valley grasslands are classified as having a Mediterranean climate of hot, dry summers and mild, rainy winters (Pitt and Heady 1978). Approximately two-thirds of the annual precipitation falls between December and March (Pitt and Heady 1978). Annual rainfall within the valley grasslands distribution ranges from 12 cm in the south to over 100 cm in the north (Bartolome et al. 2007). The seasonality of precipitation influences plant production with the greatest growth occurring during the cooler, rainy months and most plants become dormant during the hot, dry months (Eviner 2016). Climate and variability in annual precipitation have also been shown to be the main driving force influencing grassland species composition within a region (Jackson and Bartolome 2002).

Valley grasslands provide vital ecosystem services including wildlife habitat, pollinator habitat for agriculture, water filtration, and forage production (Eviner 2016). Grasslands throughout California provide critical habitat for 75 federally-listed species, including 51 plants, 14 invertebrates, and 10 vertebrates (Jantz et al. 2007). Valley grasslands contain rare ecosystem types such as vernal pools and serpentine grasslands that are biodiversity hotspots of native plants (Harrison 1999, Zedler 2003). Wild, native pollinators provide between \$937 million and \$2.4 billion worth of pollination services per year to California's agricultural industry (Chaplin-Kramer et al. 2011). These wild pollinators often rely on valley grasslands adjacent to farms (Chaplin-Kramer et al. 2011). Valley grasslands also play a role in the water filtration of pathogens and nutrients, acting as an important buffer between ecosystems and water resources (Atwill et al. 2006, Tate et al. 2006). Additionally, grassland soils tend to have high infiltration rates and can reduce the amount of stormwater runoff (Dahlgren et al. 2001). The production of forage for livestock is a direct economic benefit obtained from grasslands (Shaw et al. 2011). Approximately 4.4 million hectares of grasslands within the state provide at least 50% of the forage necessary to support livestock (FRAP 2018, Shaw et al. 2011). Rangeland quality is becoming diminished due to the threat of invasive plants reducing the amount of desirable forage species present (DiTomaso 2000).

Valley grasslands are currently dominated by European annual grasses that complete their life cycle within one year and rely on seed production for their survival (D'Antonio et al. 2007). Non-native annual grasses that have been long established in valley grasslands include wild oats (*Avena fatua*), soft brome (*Bromus hordeaceus*), ripgut brome (*Bromus diandrus*), mouse barley (*Hordeum murinum*), and rattail fescue (*Festuca myuros*) (Nafus and Davies 2014). Beginning in the early to mid-1900s, another wave of non-native plants to invade valley grasslands included non-grass species, such as yellow starthistle (*Centaurea solstitialis*), and additional invasive European grasses such as medusahead (D'Antonio et al. 2007). A recent survey of California land managers revealed that medusahead is the second-worst invasive plant to manage (Li et al. 2020). However, medusahead remains one of the least studied invasive species despite being a problematic species to manage (Li et al. 2020).

2.2 Ecology of Medusahead

Medusahead is an annual grass that can grow between 20 to 70 cm tall (Baldwin et al. 2012). In the U.S., medusahead occurs in areas that receive between 25 to 100 cm of annual precipitation (Nafus and Davies 2014). Medusahead seeds germinate from October to November with the first events of precipitation (Kyser et al. 2014) but can continue to germinate throughout winter and spring in valley grasslands (Young 1992). Vegetative growth starts in the fall with growth slowing down during the colder winter months (Sharp et al. 1957). Growth resumes in the spring with flowering occurring by June or July (Sharp et al. 1957). Medusahead matures two to four weeks later most than other annual grasses found in the western U.S. (Young 1992). New infestations of medusahead are easier to identify during this time frame because they will often be the only green plant in a sea of straw (Young 1992).

Seed dispersal begins in late July, peaks in August, and can continue into October (Davies 2008). Medusahead seed heads have long awns, or bristle-like appendages, that facilitate seed dispersal (Figure 3) (Monaco et al. 2005). Wind dispersal can carry seeds up to 0.5 m from the parent plant (Davies 2008). However, the most prominent dispersal method is by the attachment of the awns to vehicles and animals, which can cause infestations near roads and animal trails (Davies et al. 2013). Additionally, grazing cattle have been shown to disperse medusahead seeds up to 160 m (Chuong et al. 2016).



Figure 3: Medusahead seed head with visible awns (Lavin ©2007)

Medusahead is considered an invasive species in California and several other western states (Cal-IPC 2017). The California Invasive Plant Council (Cal-IPC) gives medusahead a rating of high; the organization's highest rating (Cal-IPC 2017). Medusahead is native to the Mediterranean Basin and occurs in Spain, Portugal, southern France, and Morocco (Young 1992). The first known record of medusahead in the U.S. occurred in Roseburg, Oregon in 1887 (Young 1992). Medusahead quickly spread to other states with herbarium specimens first being collected in 1901 in Washington and 1908 in California (Young 1992). Today, medusahead occurs throughout many western states including Arizona, California, Idaho, Montana, Nevada, Oregon, Utah, and Washington (Kyser et al. 2014). It is estimated that medusahead occupies at least 950,000 hectares in the western U.S. and has the potential to spread at a rate of 12% per year (Duncan et al. 2004). Within California, it is estimated that medusahead occupies 400,000 hectares and occurs in grasslands, oak woodlands, and chaparral habitats (Kyser et al. 2014). Medusahead observations have been reported and medusahead occupies extensive portions of northern and central California (Figure 4) (Calflora 2020). More recently, observations have also been reported in southern California (Calflora 2020). There have not been comprehensive statewide surveys, therefore it is likely that medusahead has a greater range than what has been incidentally observed and reported.

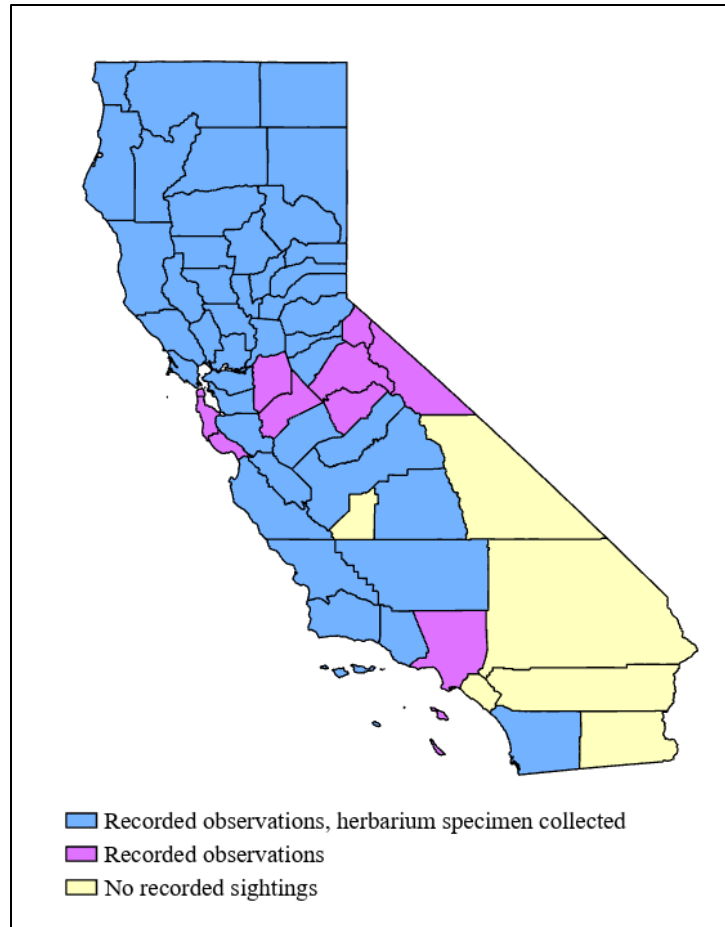


Figure 4: Medusahead distribution in California (Calflora 2020)

2.3 Impacts of Medusahead

The control of medusahead is imperative because medusahead reduces plant diversity, reduces the quality of forage, and can alter fire regimes and nutrient cycling through the accumulation of dead plant material called thatch (Kyser et al. 2014). In many areas the infestation of medusahead is advanced enough that eradication is no longer an option (Nafus and Davies 2014). Instead, controlling and preventing the spread of medusahead occurs on an annual basis by land managers of grasslands found in both public and private lands (Kyser et al. 2014).

2.3.1 Loss of Biodiversity

Medusahead can outcompete and displace many native and non-native grassland species (Davies 2011). Most studies examining the impact of medusahead on native vegetative communities are from sagebrush steppe ecosystems of the Intermountain West (Davies and Sheley 2011, Schantz

et al. 2019, Sheley et al. 2007). For example, a comparison of sagebrush sites that were invaded with medusahead and sites that were not invaded found that native plant species in the noninvaded plots had a significantly higher percent cover, density, biomass, and species richness (Davies and Svejcar 2008). Additionally, it has been observed that medusahead invasions can progress to the point where it forms monotypic stands (Kyser et al. 2014).

2.3.2 Forage Decline

A dense stand of medusahead can reduce the grazing capacity of rangelands between 50 and 80% (Hironaka 1961). Grazers typically forage on medusahead during the first two to four weeks of its growth when it is palatable (Kyser et al. 2014). However, after medusahead starts to develop flowers grazers will avoid foraging on the plant (Brownsey et al. 2017). Foraging avoidance is due to the reduced palatability of medusahead and is attributed to its high silica content and the formation of sharp awns on its seed heads (Kyser et al. 2014). In addition to grazers, seed-eating birds (Goebel and Berry 1976) and rodents (Longland 1994) tend to avoid medusahead seeds in favor of other seeds from other species.

2.3.3 Thatch Accumulation

The high amount of silica in medusahead causes it to decompose slower than other grasses (Bovey et al. 1961). Over time, a layer of medusahead thatch will accumulate (Figure 5) (Bovey et al. 1961). The reduction in decomposition rate can alter nutrient cycling by tying up nutrients in the thatch layer (Kyser et al. 2014). A thick layer of thatch reduces the amount of light penetrating to the soil surface and inhibits the recruitment and germination of native species (Mariotte et al. 2017). The thatch layer increases seed production of medusahead plants and increases seed germination of wild oats (Mariotte et al. 2017). It is thought that plant species with smaller seed sizes (e.g., forbs and some native grasses) do not have enough stored energy to create a shoot system long enough to penetrate through the thatch layer (Kyser et al. 2014). The thatch layer creates a positive feedback loop by altering the local growing conditions to favor non-native grasses while preventing the growth of native species (Coleman and Levine 2007). Medusahead thatch also alters the local fire regime (Nafus and Davies 2014). The thatch layer can increase the frequency of fires due to a buildup of horizontal fuel across a landscape (Brooks et al. 2004). In areas with monotypic stands of medusahead, the lack of a break in the thatch layer can increase the footprint of a potential fire (Brooks et al. 2004).



Figure 5: Medusahead thatch (Lavin ©2003)

2.3.4 Economic Cost

The economic cost of managing medusahead is not mentioned in published studies. A meta-analysis of 22 studies focusing on medusahead control reported that zero of the studies included the cost of medusahead management (James et al. 2015). However, one example is a 0.8-hectare infestation that costs approximately \$2,000 per year to manage (Voeller, Point Reyes National Seashore, personal communication). The total cost includes the herbicide Milestone, staffing, and equipment rental to apply the herbicide. Additionally, reseeding an area after a medusahead management treatment is often cost-prohibitive (Kyser et al. 2014). Native grass and forb seed and nursery plugs can thousands of dollars per hectare, which does not include the cost of labor or planting equipment.

2.4 Medusahead Control Strategies

Land managers have multiple strategies in their arsenal to control medusahead infestations. Management strategies can be categorized as mechanical, chemical, cultural, or biological (Kyser et al. 2014). Mechanical control encompasses hand pulling, mowing, tilling, and other forms of mechanical removal of above-ground biomass (DiTomaso et al. 2010). Chemical control includes the use of herbicides in pre- or post-emergent treatments (DiTomaso et al. 2010). Pre-emergent treatments are applied to the soil in the fall to prevent seeds from

germinating (Kyser et al. 2014). Post-emergent herbicides are applied in the spring to kill a plant after vegetative growth already has begun (Kyser et al. 2014). Cultural control encompasses targeted grazing, prescribed burns, and revegetation or reseeding after control treatments have occurred (Nafus and Davies 2014). Biological control is used by introducing a natural predator or disease to assist in reducing the biomass of an invasive plant (Nafus and Davies 2014). In the U.S. there are no commercially available biological control methods being utilized to manage medusahead (Nafus and Davies 2014).

3. Methods

A comprehensive literature review was conducted to facilitate a comparative analysis of the treatment literature for medusahead. The search terms “caput-medusae” or “medusa*” and “California” were utilized to obtain peer-reviewed papers from the following databases: FUSION, SCOPUS, Environment Complete, and ProQuest Dissertations & Theses Global. All results were examined to determine if they were relevant to this study. The same search terms were utilized in Google Scholar and the first 100 papers were analyzed for relevancy. Relevant papers were identified using the following criteria: medusahead being a target species, study location within the valley grassland range, the inclusion of one of the four control strategies being used on medusahead, and the inclusion of data on non-target plant species. Only articles that met all four criteria were included in this study. The bibliographies of identified studies were utilized to locate additional potential studies using the same criteria.

A synthesis table was utilized for the comparative analysis. For each study identified in the literature search, I recorded the following information: study location, length of study, plot size, data collection metrics, treatment type, treatment impacts on medusahead, and treatment impacts on non-target plants. An abbreviated version of the synthesis table is presented in the results.

4. Results

A total of 14 medusahead control studies were identified and are summarized in Table 1. Of these studies 3 used herbicides, 4 used grazing, 6 used prescribed burning, and 2 used mechanical control. One study was introduced as using a gradient of grazing intensities (Stein et al. 2016). However, I classified this study as using mechanical control. The grazing gradient was achieved through mowing, not grazing. The metrics used on non-target plant species in these studies were percent cover (n=13), species richness (n=4), species composition (n=2), Shannon diversity index (n=2), vigor (n=1), germination (n=1), and fecundity (n=1).

Table 1: Synthesis table

Reference	Metrics	Treatment				Results
		Herbicide	Grazing	Burning	Mechanical	
Kyser et al. 2012	Percent cover	X (aminopyralid, rimsulfuron, imazapic)				<p>Aminopyralid - significantly decreased or eliminated forb cover; significantly increased annual grass cover</p> <p>Rimsulfuron - no significant effects on forbs or annual grasses</p> <p>Imazapic - an increasing trend in forb cover, no significant impacts on annual grass cover</p>
Kyser et al. 2007	Vigor Percent cover Species richness	X (imazapic)				<p>Selectivity trials – pre-emergent (PRE) application caused greater overall damage than post-emergent (POST) application; forbs were highly tolerant to PRE treatments and moderately to highly tolerant to POST treatments; native perennial grasses showed a variable response with tolerances ranging from low to moderate</p> <p>Rate trials (PRE application) - annual grass and forb cover decreased with an increasing rate of imazapic; annual grass cover was lower in disturbed plots; forb cover was not impacted by disturbance</p> <p>Species richness - Yuba Co. site had no significant difference on species richness; Yolo Co. site had a significantly higher species richness in disturbed plots; plots with higher application rates tended to have lower species richness</p>
Rinella et al. 2018	Percent cover	X (aminopyralid)				<p>Forage grasses - fall treatments ranged from no impact to increased cover; spring treatments showed increased cover for all three forage grass species with increases dependent upon species; soft brome (<i>Bromus hordeaceus</i>) had the highest increase in cover</p> <p>Forbs – fall and spring treatments decreased cover of forbs when present in the treatment plots</p> <p>Other non-native grasses - spring and fall treatments caused slight increases in cover at two out of five of the sites</p>

Reference	Metrics	Treatment				Results
		Herbicide	Grazing	Burning	Mechanical	
Davy et al. 2015	Percent cover		X (cattle)			<p>Grazing – significant increase of cover for filaree (<i>Erodium</i> spp.) and slender oat (<i>Avena barbata</i>); significant decrease of cover for red brome (<i>Bromus madritensis</i>) and ripgut brome (<i>Bromus diandrus</i>)</p> <p>Ungrazed - significant increase of cover for red brome, ripgut brome, and slender oat; significant decrease of cover for filaree</p>
Reiner and Craig 2011	Percent cover Species richness		X (cattle)			<p>No significant difference in species richness or percent cover between grazed and ungrazed plots</p> <p>Approximately 60% of 170 identified plants were native, but 81% of the total cover consisted of non-native species</p>
James et al. 2017	Species composition		X (cattle, sheep)			<p>Cattle – Native plants had a positive response ratio (i.e., increase) in both treatments; desirable forage had a neutral response in the low density/long duration treatment and a negative response ratio (i.e., decrease) in the high density/short duration treatment</p> <p>Sheep – A neutral response ratio was observed for native plants and desirable forage in the low density/long duration treatment and for desirable forage in the high density/short duration treatment; native plants in the high density/short duration treatment had a positive response ratio</p>
DiTomaso et al. 2008	Percent cover Species richness Shannon diversity index		X (sheep)			<p>Annual grass – early + midspring (March + April/May) grazing had a significant decrease in cover 1 year post-grazing; the significant difference between treatments disappeared 2 years post-grazing</p> <p>Forbs - midspring grazing treatment significantly increased native and non-native forb cover</p> <p>Species richness and diversity – early + midspring grazing significantly increased species richness and the Shannon diversity index both 1- and 2-years post-grazing</p>

Reference	Metrics	Treatment				Results
		Herbicide	Grazing	Burning	Mechanical	
Kyser et al. 2008	Percent cover Shannon diversity index			X		Vegetative cover – forb cover significantly increased; unpalatable forb cover did not change significantly; significant decreases in other grasses were observed after the first burn, but not after the second burn Species diversity - no significant changes were observed
Davy and Dykier 2017	Percent cover			X		Forbs - Filaree increased from 4% to 65%, 55%, and 55% (1, 2, and 3 years post-burn, respectively) Native plants - No significant change in native plant composition between burned and unburned plots
Pollack and Kan 1998	Percent cover			X		Native species - grasses significantly increased in cover in upland and transition habitats; forbs significantly increased in transition and vernal pool habitats Non-native species - grasses significantly decreased in cover in upland and transition habitats; forbs significantly increased in all three habitat types
Marty 2015	Percent cover Species composition			X		Non-native forbs - cover increased almost 100% 1 year post-burn in upland plots, increase not significant 2 years post-burn Native forbs - no significant change in cover; richness increased by 15% after 1 year post-burn Native perennial grasses - significant increase in cover 1 year post-burn (cover was 1.8 times higher than unburned plots) Species composition – no significant change
Betts 2003	Percent cover Species richness			X		Native species – not analyzed due to low frequency Non-native grasses – Soft brome cover decreased; rattail fescue cover increased with consecutive burns having the greatest increase; neutral or minimal impacts on slender oat, ripgut brome, and Italian ryegrass Non-native forbs – Rose clover (<i>Trifolium hirtum</i>) and filaree cover increased but impacts were short-lived Species richness - No significant impact on total plant cover or species richness

Reference	Metrics	Treatment				Results
		Herbicide	Grazing	Burning	Mechanical	
Berleman et al. 2016	Germination Fecundity Percent cover			X	X	<p>Germination – Wild oats had a 26% germination in burned plots compared to 80% germination in unburned plots; no purple needlegrass (<i>Stipa pulchra</i>) seeds were observed in either treatment</p> <p>Fecundity - burning had a significant increase of wild oats; burning had a trending decrease of purple needlegrass</p> <p>Transitions between vegetation types – 29 observed vegetation shifts occurred; 17 of the shifts resulted in a filaree dominated plot</p>
Stein et al. 2016	Percent cover				X	<p>Vegetative transitions (categories: medusahead, native perennial, annual forage, exotic forb) – 304 total transitions occurred; 7 plots never transitioned; 3 transitions from medusahead to native perennial; transitions from native perennial to medusahead (14), annual forage (7), and exotic forb (30)</p> <p>Mowing intensity - all vegetation states transitioned to exotic forbs under high-intensity mowing; no impact on the transition between any of the three grass assemblages</p>

4.1 Herbicides

Commercial production of synthetic herbicides began in the 1940s for the control of agricultural weeds (Timmons 2005). By the 1970s and 1980s, companies were able to produce a variety of herbicides with different biochemical properties that could target specific groups of plants; these targeted herbicides are differentiated by their mode of action on the various plant groups (Appleby 2005). While herbicides were invented for use in agricultural fields, they have proven to be a useful tool for managing invasive plants in natural ecosystems (Wagner et al. 2017). The differing goals of herbicide use in agriculture and natural areas have led to two different approaches to herbicide application (Crone et al. 2009). In agricultural settings, herbicides are applied broadly because they are used to kill all other plants except the crop plant (Crone et al. 2009). In natural ecosystems, the goal of herbicide use is to kill a specific plant while minimizing impacts to the remainder of the ecological community (Crone et al. 2009). However, little research has been conducted on the non-target effects of herbicides, especially on native plants (Wagner et al. 2017). Land managers must weigh the harm caused by an invasive plant versus the harm caused by herbicide application and the unknown impacts that herbicides may have on the entire ecosystem (Skurski et al. 2013).

There are ways to apply herbicides that reduce non-target impacts and lessens the risk of damaging other plants. Herbicide efficacy is influenced by the method and rate of application as well as the growth stage and morphology of the target plant (DiTomaso and Smith 2012). Broadcast applications over large areas are achieved with aerial or ground vehicles and more are common in agricultural settings or dense infestations of an invasive plant (Kyser et al. 2014). Spot treatments are more common within natural areas and require greater application precision that is achieved with a backpack sprayer (Kyser et al. 2014). The use of spot treatments is an effective way to control the quantity and the application rate of herbicides to a target plant while reducing impacts to non-target plants (Power et al. 2013).

The active ingredients utilized in studies on medusahead control in valley grasslands included aminopyralid, imazapic, and rimsulfuron. Another common herbicide that is used to control medusahead is glyphosate (Kyser et al. 2013). However, no studies located in valley grasslands used glyphosate. These four herbicides have different selectivity, application timing, and modes of action (Table 2).

Table 2: Characteristics of herbicides utilized in medusahead control studies

Active Ingredient (Herbicide)	Application Rates in Studies	Selectivity	Timing of Application	Mode of Action
Aminopyralid (Milestone)	53, 55, 88, 123, 245 g/ha	Forb plants	Pre- and Post-emergence	Growth regulator
Imazapic (Plateau, Panoramic 2SL)	0, 35, 70, 105, 140, 175, 210, and 280 g/ha	Nonselective	Pre- and Post-emergence	Amino acid synthesis inhibitor
Rimsulfuron (Matrix)	18 and 35 g/ha	Nonselective	Pre- and Post-emergence	Amino acid synthesis inhibitor
Glyphosate (Roundup)	n/a	Nonselective	Post-emergence	Amino acid synthesis inhibitor

(Sources: Kyser et al. 2007, Kyser et al. 2013, Rinella et al. 2018, DiTomaso and Smith 2012)

4.1.1 Advantages

Herbicides are an effective management strategy for controlling invasive plants because they are typically cheaper, they require less human labor, and the control of invasive plants can occur over a shorter time frame (Wagner et al. 2017). Additionally, modern herbicides are generally less toxic to the environment than first-generation herbicides (Wagner et al. 2017). Herbicides used in natural areas are typically water-soluble and quickly degrade in the environment (Tatum 2004). These herbicides are also less likely to pose a risk to fish and wildlife because herbicides are designed to target plant biochemical processes (Tatum 2004).

Compared to grazing and prescribed burning, herbicides are the most effective management strategy to control medusahead (James et al. 2015). Higher rates of herbicide application provide a greater reduction of medusahead cover (Figure 6) (Kyser et al. 2012). Pre-emergent application of aminopyralid in the fall at a rate of 245 g/ha reduced medusahead cover by an average of 89% (Kyser et al. 2012). Post-emergence application of aminopyralid in the spring was found to

nearly eliminate medusahead cover (Rinella et al. 2018). Herbicides are also the recommended management strategy for medusahead removal if a site is to be revegetated or restored (Kyser et al. 2007).

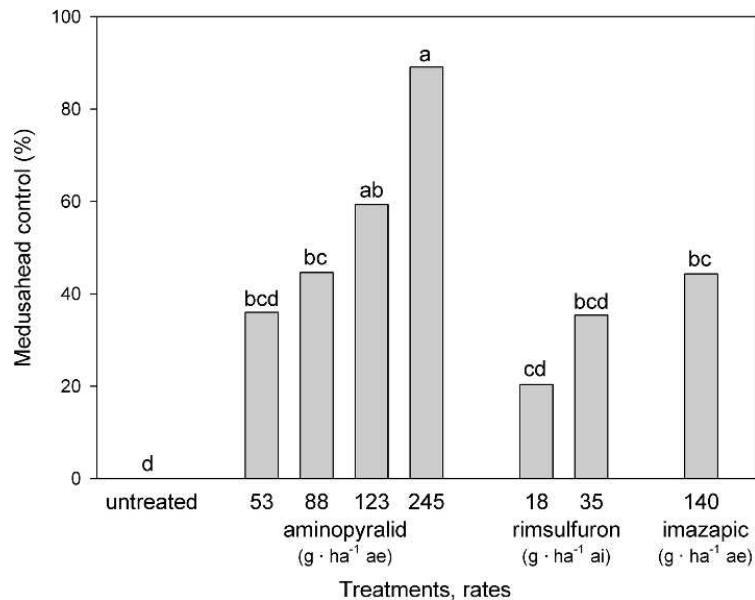


Figure 6: Medusahead control with the pre-emergent application of the herbicides aminopyralid, rimsulfuron, and imazapic. Bars with the same letter (a, b, c, or d) are not significantly different ($\alpha = 0.05$) (Kyser et al. 2012).

4.1.2 Disadvantages

Herbicides that are applied to a plant often do not remain in the same location. Herbicides can move to non-target areas through an assortment of mechanisms (Egan et al. 2014). The most common mechanisms include spray drift (wind dispersal), vapor drift (air dispersal), and dispersal in surface or subsurface water flow (Egan et al. 2014). Spray drift of the herbicides dicamba and glyphosate has been shown to reduce seed production potential in native plants (Olszyk et al. 2017). Additionally, herbicides can alter species abundance and composition of plant communities while having minimal long-term impacts on the target invasive plant (Freemark and Boutin 1995).

In addition to impacting plants, herbicides have the potential harm to human health. Chronic exposure to certain herbicides has been shown to increase the risk of certain cancers and neurologic disorders (Alavanja et al. 2004). The public perception of herbicide use is often negative and leads to conflicts with public agencies and land managers striving to use herbicides to control invasive plants (Norgaard 2007). In some instances, public perception can lead to herbicides being banned. For example, the Marin Municipal Water District opted to remove herbicides from their vegetation management plans in response to growing concerns from the community about the public health risks associated with herbicide exposure (Marin Municipal Water District 2019).

4.1.3 Impacts on Plant Community

The three studies on herbicide application to treat medusahead did not report on the non-target impacts on native species (Rinella et al. 2018, Kyser et al. 2012, Kyser et al. 2007). The lack of analysis is likely due to the absence or low frequency of native species at the three study sites. However, both native and non-native plant species were utilized in a selectivity trial of imazapic (Kyser et al. 2007). The selectivity trial was used to determine the tolerance level of thirty species that were planted as seeds into prepared beds.

The impacts of herbicide use on non-native annual grasses depended upon the specific herbicide being used. Pre-emergent application of aminopyralid resulted in variable impacts on non-native annual grasses across all application rates (Kyser et al. 2012, Rinella et al. 2018). One study observed a significant increase in the percent cover of non-native annual grasses across all application rates of aminopyralid (Kyser et al. 2012). Another study saw neutral to increased non-native annual grass cover, likely due to the timing of herbicide application and low precipitation (Rinella et al. 2018). Post-emergent application of aminopyralid resulted in a decrease in seed production and an increase in percent cover of non-native annual grasses (Rinella et al. 2018). The pre-emergent use of imazapic and rimsulfuron resulted in no significant change to the percent cover of non-native annual grasses compared to untreated plots (Kyser et al. 2012). Lastly, non-native annual grass cover decreased with an increasing rate of imazapic (Kyser et al. 2007).

The selectivity trial of three rates of imazapic (70, 140, and 280 g/ha) supports the idea that this herbicide tends to target some non-native grasses (Kyser et al. 2007). Imazapic was applied either soon after seeding (i.e., pre-emergent) or the following spring (i.e., post-emergent) (Kyser et al. 2007). Native perennial grasses were found to be slightly more tolerant to imazapic at all three rates and both pre- and post-emergent treatments than non-native annual grasses (Kyser et al. 2007). The native perennial and non-native annual grasses showing the highest tolerance to imazapic belonged to species in the genera *Hordeum* (barley) and *Elymus* (wheatgrass) (Kyser et al. 2007). The authors provided no details to explain why these specific genera would be more resistant to imazapic. Additionally, four native forb species and yellow starthistle all showed high tolerance of imazapic in both the pre- and post-emergent treatments (Kyser et al. 2007).

The impacts of herbicides on non-native forb cover also depended upon which herbicide was used. All pre-emergent application rates of aminopyralid resulted in a drastic reduction in non-native forb cover, with many sites seeing the complete elimination of forb plants (Kyser et al. 2012, Rinella et al. 2018). Pre-emergent application of imazapic increased non-native forb plant cover, while rimsulfuron had no significant impact on forb plant cover (Kyser et al. 2012). Conversely, another study found that non-native forb cover decreased with increasing imazapic rate (Kyser et al. 2007).

4.2 Grazing

Grazing for conservation purposes has a tumultuous record in California (Krausman et al. 2009, Jensen 2001). Proponents argue that responsible grazing can help manage wildfire fuel levels, enhance nutrient cycling, reduce invasive plants, increase native plants, and support the economy of California (DiTomaso and Smith 2012). On the other hand, grazing practices can detrimentally impact species sensitive to grazing, increase soil erosion and compaction, and pollute and damage riparian and wetland habitats (Fleischner 1994). Despite these differences, grazing is the most common management strategy used by land managers to manage invasive plants (James et al. 2015).

A theoretical approach to managing invasive annual grasses in a grassland ecosystem is called the “Green and Brown” grazing strategy (Figure 7) (DiTomaso and Smith 2012). The strategy manipulates the timing and intensity of grazing based on the differences in life-history stages between annual and perennial grasses (DiTomaso and Smith 2012). Native plants in valley grasslands tend to be long-lived perennial bunchgrasses, whereas invasive plants tend to be annual grasses (Menke 1992). In the winter, perennial grass species tend to be brown, dormant, and less vulnerable to grazing (Frost and Launchbaugh 2003, DiTomaso and Smith 2012). On the other hand, annual grass species are starting to germinate and are the dominant green forage species in a grassland (DiTomaso and Smith 2012). High-density grazing over a short duration should be allowed during this time frame to cause the greatest reduction in annual grasses (Menke 1992). As the season progresses, grazers should be removed from the grasslands to prevent damaging perennial grasses (Menke 1992).

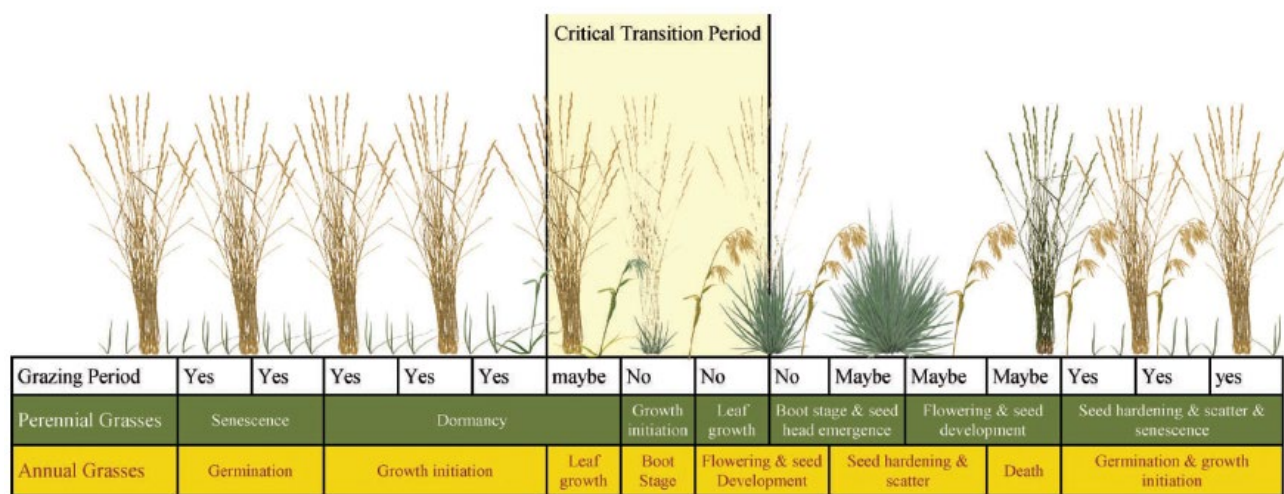


Figure 7: The Green and Brown grazing strategy. High-density grazing should be applied during the window of time where the shorter, perennial grasses are dormant and the taller, annual grasses are beginning to grow. Grazers should be removed during the critical transition period to avoid grazing upon perennial grasses (DiTomaso and Smith 2012).

4.2.1 Advantages

Grazing as a management strategy to control invasive plants is highly flexible due to the ability to manipulate grazing density and timing to achieve a desired goal. Furthermore, land managers can utilize the differences in foraging behaviors and preferences between cows and sheep (DiTomaso et al. 2010). To control medusahead with sheep, the optimal timing is typically during late spring before medusahead has a chance to produce seed heads (Figure 8) (DiTomaso et al. 2008). Similarly, the optimal timing for cattle grazing is in late spring when medusahead is more palatable to cattle (Davy et al. 2015). The greatest reduction of medusahead was observed when grazers, both cattle and sheep, were stocked at a high density over a short duration (James et al. 2017).

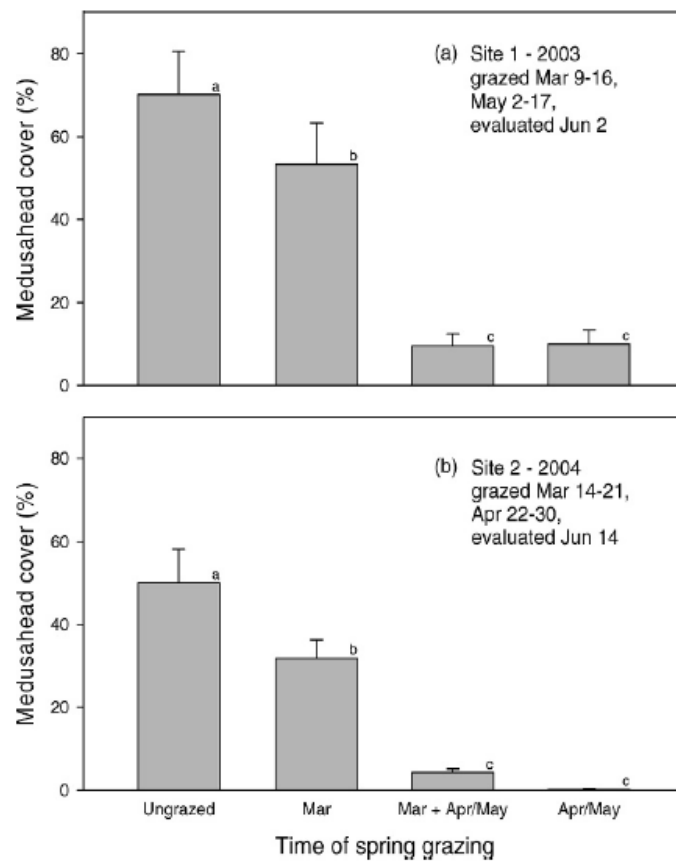


Figure 8: Comparison of spring timing of sheep grazing on medusahead cover. The duration of grazing treatments occurred in early spring (Mar), late spring (Apr/May), or through the entire spring (Mar + Apr/May). Error bars represent one standard error. Values with different letters are significantly different ($p = 0.05$) (DiTomaso et al. 2008).

Another benefit of grazing is the reduction in thatch levels (DiTomaso et al. 2008). Grazers consume aboveground biomass preventing the accumulation of thatch (James et al. 2017). It is also thought that trampling by grazers, especially cattle, facilitates the breakdown of thatch (George et al. 1989). The reduction of thatch creates opportunities for other plant species to germinate due to an increase in bare ground and an increase in the amount of light reaching the soil surface (Reiner and Craig 2011).

4.2.2 Disadvantages

Grazers, especially at high densities, can impact soil structure and diminish water quality (Kauffman and Krueger 1984). A common effect of grazing is the compaction of soil (Byrnes et al. 2018). Compacted soil has reduced water infiltration capability leading to greater surface runoff and the potential for greater soil erosion (Emmerich and Heitschmidt 2002, Byrnes et al. 2018). Grazers tend to congregate near shady areas and water sources, such as riparian areas (Kauffman and Krueger 1984). As a result, it is expected that these areas would have higher levels of soil compaction and erosion. Additionally, water runoff carries fecal matter and sediments into streams and rivers causing a reduction in water quality (Kauffman and Krueger 1984).

The control of medusahead with grazing has variable results. Reduction in medusahead cover is temporary and has different impacts depending on the timing of grazing (Nafus and Davies 2014). Fall and early spring grazing had no impact on controlling medusahead (DiTomaso et al. 2008). In some sites, year-round grazing can increase medusahead cover (Harrison et al. 2003). Furthermore, the percent cover of medusahead in ungrazed plots can be variable and it becomes difficult to untangle the impacts that grazing and precipitation have on medusahead cover (Figure 9) (Davy et al. 2015, DiTomaso et al. 2008).

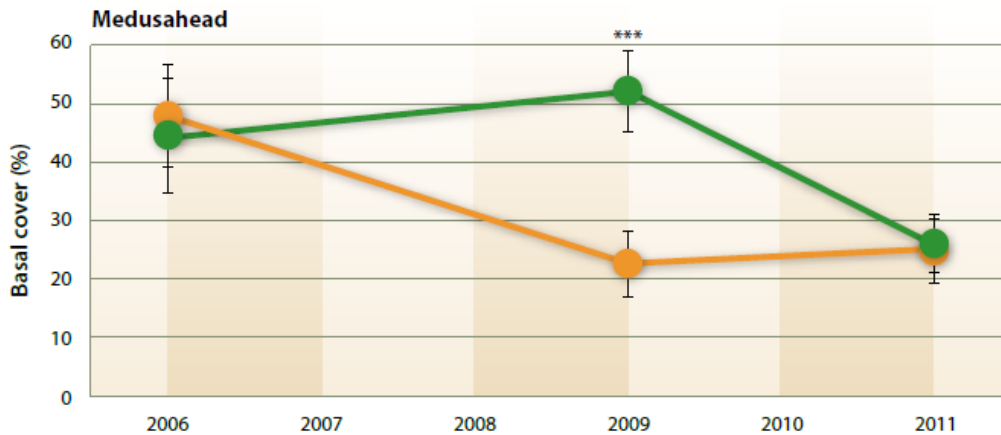


Figure 9: Comparison of medusahead cover from grazed and ungrazed plots. Orange indicates grazed, green indicates ungrazed. Error bars are equal to one standard error. Asterisks indicate significance at $p < 0.01$ (Davy et al. 2015).

4.2.3 Impacts on Plant Community

Grazing for medusahead control generally resulted in decreases to non-native annual grasses and increases in non-native forbs. Cattle grazing resulted in a significant increase in percent cover of filaree (*Erodium* sp.) and slender oat (*Avena barbata*) and a significant decrease in ripgut brome and red brome (*Bromus madritensis*) (Davy et al. 2015). In the same study, a significant increase of percent cover of slender oat, ripgut brome, and red brome was observed in ungrazed plots (Davy et al. 2015). The increase of these non-native grasses contributed to the decrease of medusahead in both the grazed and ungrazed treatment plots (Davy et al. 2015). At another study site, cattle and sheep grazing were found to have neutral impacts on non-native forage plants at both higher and lower grazing densities (James et al. 2017). Sheep grazing occurring throughout early and midspring (March – May) resulted in a significant decrease in annual grasses (DiTomaso et al. 2008). Midspring (April – May) grazing with sheep resulted in a significant increase of forb cover with cover increasing nearly three times that found in the ungrazed plots (DiTomaso et al. 2008).

Grazing has variable impacts on native plants. A five-year grazing study conducted on five different working ranches in northern California found no significant difference in native species richness nor percent cover between grazed and ungrazed plots (Reiner and Craig 2011). Conversely, another study found cattle grazing had a positive response (i.e., increased percent cover) on native plants at two different grazing prescriptions: low density over a longer time and

higher density over a shorter time (James et al. 2017). With sheep grazing, the higher density grazing treatment resulted in a more positive response than the lower density grazing (Figure 10) (James et al. 2017). Differences in location and historical land management practices likely resulted in a different response of native plants to grazing. However, James et al. (2017) did not provide background details of grazing practices of their study sites to draw any conclusions.

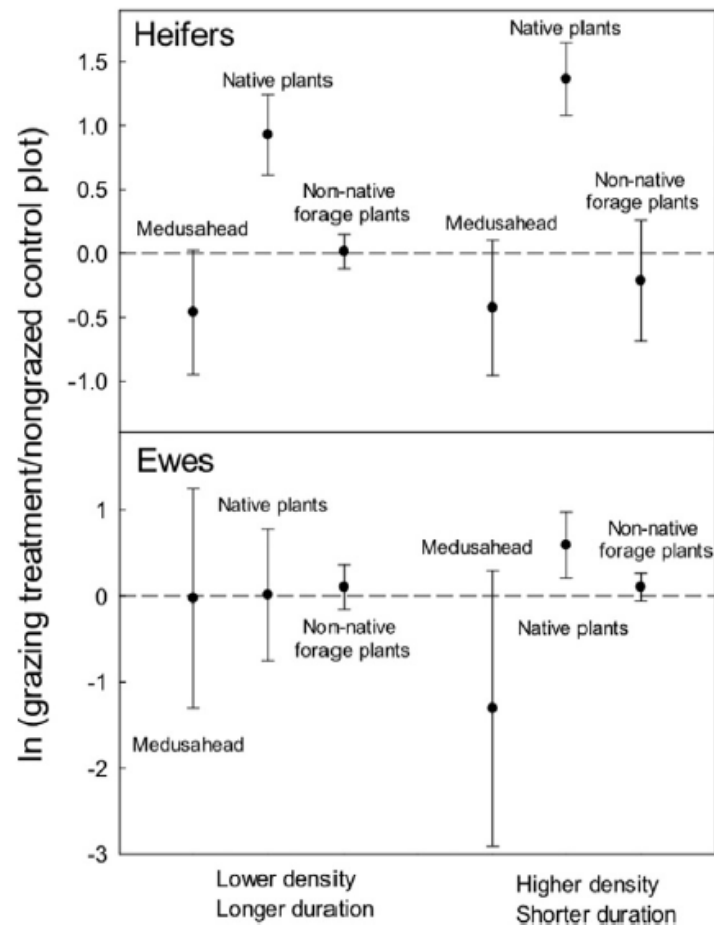


Figure 10: Comparison of different grazing densities and timing on response ratios of medusahead, native plants, and non-native forage plants. Heifers = female cattle and ewes = female sheep. Solid circles are the mean. Error bars are the 95% confidence interval (James et al. 2017).

Of the four studies that examined controlling medusahead with grazing, only one reported on the impact of species richness and diversity. Sheep grazing occurring during early and midspring and only during midspring resulted in significant increases in species richness (number of species per m²) and species diversity (Shannon diversity index) (DiTomaso et al. 2008). The increase in species richness was largely attributed to an increase in native forbs (DiTomaso et al. 2008).

4.3 Prescribed Burning

Prescribed burning has a long history of usage as a land management tool. Before the arrival of European settlers, Native Americans would tend their land and shape their environment with fire (Anderson 2019). Fire was used to flush wildlife from an area to facilitate hunting and to enhance the growth of beneficial plants (Anderson 2019). Today, fire is typically used to achieve specific land management goals. These goals include reducing fuel loads to lessen wildfire risk, controlling invasive plants, and to increase native biodiversity (DiTomaso et al. 2006).

Prescribed burns can result in a short-term increase in percent cover of native species while wildfires tend to increase the percent cover of non-native species (Alba et al. 2015).

Additionally, some native perennial bunchgrasses (e.g., *Festuca idahoensis* and *Stipa pulchra*) have been shown to respond positively to fire (Defossé and Robberecht 1996, Fehmi and Bartolome 2003).

4.3.1 Advantages

Prescribed burns have been shown to significantly decrease the percent cover of medusahead (Davy and Dykier 2017, Berleman et al. 2016, Kyser et al. 2008). However, these decreases are typically observed over a short time frame. At one site, medusahead cover decreased from 77% before a prescribed burn to 4% the year following the burn (Davy and Dykier 2017).

Medusahead cover increased the second and third year after the burn to 17% and 18%, respectively (Davy and Dykier 2017). Prescribed burns can also significantly reduce or eliminate the thatch layer allowing other plant species a chance to germinate (Figure 11) (Pollak and Kan 1998).

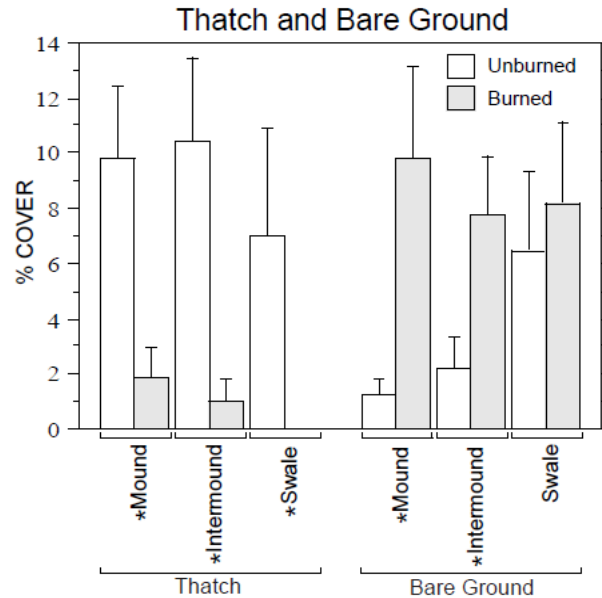


Figure 11: Comparison of percent cover of thatch and bare ground between unburned and burned plots in a vernal pool ecosystem. Mound = upland or grassland, intermound = transition zone between mound and swale, swale = vernal pool depression. Error bars are equal to one standard error. Asterisks indicate a significant difference at $\alpha = 0.1$ (Pollak and Kan 1998).

The delayed maturation of medusahead makes prescribed fire a suitable management strategy for controlling this species. Land managers can time a prescribed burn to occur during the two- to four-week window when medusahead is still maturing but other grass species have already dropped their seeds (Murphy and Lusk 1961). Seeds are more likely to be killed when they are still attached to the parent plant and can be exposed to the direct flames of the fire (DiTomaso et al. 2006). A prescribed burn can kill 90% of aboveground medusahead seeds in 4.8 to 6.5 seconds (Sweet et al. 2008). As summer progresses and medusahead starts to dry out and lose moisture, it can take as little as 1 second to kill a medusahead seed that is still attached to the parent plant (Sweet et al. 2008).

4.3.2 Disadvantages

The use of prescribed burns has decreased over the years due to public perception, smoke, and regulatory barriers (Kolden 2019). A 2008 survey of land managers found the biggest obstacles to utilizing a prescribed fire was the narrow time window allowed for burning, meeting air quality regulation, and not have enough personnel (Quinn-Davidson and Varner 2012). Although infrequent, prescribed burns can escape their planned boundaries. An example of a prescribed

burn that escaped is the Lowden Ranch fire near Redding, CA on July 2, 1999 (USDI BLM 1999). The fire was planned to burn 40 hectares of invaded grassland but wound up consuming 810 hectares and destroying 23 residences before being controlled (USDI BLM 1999).

Prescribed burns are not effective at killing vegetative biomass and seeds that reside underground. For example, medusahead seeds in the soil are not likely to be killed by a prescribed burn (Sweet et al. 2008). A prescribed burn in grassland habitat was recorded to reach temperatures between 219 and 231°C (DiTomaso et al. 1999). A burn of 250°C at the soil surface would need to last, on average, for 28.0 seconds to kill half of the medusahead seeds present in the soil (Sweet et al. 2008). Most grassland fires do not persist for 28 seconds and are therefore not an effective tool for removing medusahead from the seed bank (Sweet et al. 2008).

Prescribed burns also impact wildlife relying upon valley grasslands. Short-term impacts on small mammals due to burns include the loss of protective vegetation, increased predation, and a potential decrease in seed forage (Crownier and Barrett 1979). Ground-nesting birds are at risk of having their nest or young killed by burns occurring in the spring during their breeding season (Kruse and Piehl 1984). Invertebrates in the egg or larval life stage have a higher chance of dying during a burn than their more mobile adult forms (Kral et al. 2017).

4.3.3 Impacts on Plant Community

The most prominent impact of conducting a prescribed burn to manage medusahead is a change in the distribution of percent cover of the plant community (Kyser et al. 2008, Betts 2003). Even with a shift in percent cover, there was not a significant difference in species diversity between burned and unburned treatments (Figure 12) (Kyser et al. 2008). Similarly, a site in the Sierra Nevada foothills found no significant impact on species richness between burned and unburned plots (Betts 2003). However, one study conducted in a vernal pool grassland found significant changes in species composition (Marty 2015). It is thought that this change in species composition was due to a change in nutrient availability or a change in hydrodynamics of the vernal pool (Marty 2015).

Site	Total species	Percent vegetative cover		
		> 20%	10 – 20%	5 – 10%
Fresno				
Control	13	<i>Bromus hordeaceus</i> <i>Elymus caput-medusae</i>	<i>Erodium brachycarpum</i> <i>Festuca myuros</i>	<i>Bromus diandrus</i> <i>Hordeum murinum</i>
Burned	12	<i>Erodium brachycarpum</i> <i>Festuca myuros</i>		
Yolo				
Control	12	<i>Bromus hordeaceus</i> <i>Elymus caput-medusae</i>	<i>Avena barbata</i>	<i>Hordeum murinum</i>
Burned	10	<i>Avena barbata</i> <i>Erodium brachycarpum</i>	<i>Festuca myuros</i>	<i>Bromus hordeaceus</i>

Figure 12: Total number and dominant species at two study sites following the second year of prescribed burning. Bold indicates grass species (Kyser et al. 2008).

Most studies observed an increase in forb cover after burning. The most common forb in these studies was filaree. At one site, filaree increased from 4% cover in the pre-burn survey to 65% cover one year after a prescribed burn (Davy and Dykier 2017). Another site observed increased cover of both filaree and another forb, rose clover (*Trifolium hirtum*) (Betts 2003). At another study site, transformations from one dominant vegetative type to another were observed 17 times in burned plots (Berleman et al. 2016). Of the observed vegetative transitions, 12 of the transitions were from medusahead to filaree and 3 were from needlegrass mix to filaree (Berleman et al. 2016). In vernal pool grasslands, a similar pattern was observed with non-native forbs significantly increasing in cover after a burn (Figure 13) (Marty 2015, Pollak and Kan 1998). One study found non-native forb percent cover increased by 100% in burned plots (Marty 2015). However, the difference between burned and unburned plots was not detectable two years after the burn (Marty 2015). Another vernal pool study found a significant increase in non-native forb cover in the upland grassland plots and attributed the increase in cover to filaree (Pollak and Kan 1998).

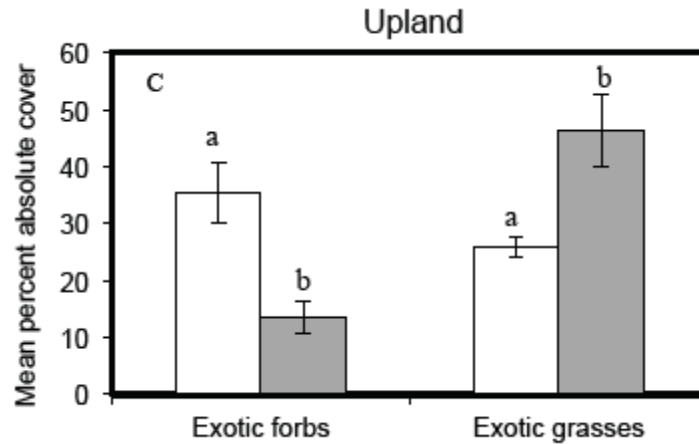


Figure 13: Comparison of percent cover of burned and unburned plots. White bars = burned plots, gray bars = unburned plots. Error bars equal to one standard error. Bars with different letters are significantly different ($p < 0.05$) (Marty 2015).

Prescribed burns resulted in an overall decrease in percent cover of non-native annual grasses (Kyser et al. 2008). The Sierra Nevada foothill study site observed mixed impacts to percent cover with a decrease in soft brome cover and an increase in rattail fescue cover (Betts 2003). No change to slight decreases in percent cover were observed in slender oats, ripgut brome, and Italian ryegrass (*Festuca perennis*) (Betts 2003). Impacts on reproduction were also observed in wild oats with a decrease in their germination rate from 80% to 26% in burned plots (Berleman et al. 2016). Burning had a significant increase in wild oat fecundity, with fecundity measured as the number of glumes (i.e., flowers) (Berleman et al. 2016). In vernal pool grasslands, the percent cover of non-native grasses was significantly reduced by 35% one year after a burn (Marty 2015). Similarly, another vernal pool grassland site observed a 40% reduction in non-native grasses (Pollak and Kan 1998).

Prescribed burns have mixed impacts on native plant species. Burning had a slightly negative impact on purple needlegrass (*Stipa pulchra*) fecundity, with fecundity measured as the number of flowering stems (Berleman et al. 2016). A 200-hectare burn had no significant impact on the total percent cover of native wildflowers (Davy and Dykier 2017). In vernal pool grasslands, a prescribed burn had no significant impact on the total percent cover of native forbs, but species richness of native forbs increased an average of 15% (Marty 2015). At another vernal pool site, both native grasses and forbs had a significant increase in percent cover after a burn (Pollak and Kan 1998).

The impacts of prescribed burns on the plant community composition appear to be short-lived. Davy and Dykier (2017) chose to terminate monitoring their study after 3 years when there was no significant difference in species composition between the unburned and burned plots. At another site, an increase in filaree cover was short-lived and resembled the unburned plots one-year post-burn (Betts 2003). Similarly, the vernal pool grassland study sites showed the observed changes in percent cover were no longer significantly different two years post-burn (Marty 2015). The only exception was the vernal pool grassland at Jepson Prairie Preserve. The percent cover and species richness of native species remained elevated compared to control plots at the end of the three-year study period (Marty 2015). Jepson Prairie Preserve had the highest cover of non-native grasses and thatch out of four vernal pool sites (Marty 2015). The surge of native species cover is attributed to the removal of thatch and decreased pressure from non-native grasses (Marty 2015).

4.4 Mechanical control

Mechanical control refers to management strategies that completely remove or physically damages a plant to the point where it can no longer reproduce or survive (DiTomaso et al. 2010). For annual grasses and forbs, the most common mechanical control strategy is mowing (Masters and Sheley 2001). However, mechanical control is not as commonly used as other management strategies due to the high cost of implementation (Masters and Sheley 2001).

4.4.1 Advantages

Mechanical control is primarily used to decrease seed production (DiTomaso 2000). Both studies in this analysis that utilized mechanical control referred to their mechanical control treatment as seed limiting. For controlling medusahead, the optimal time for mowing is when the plant has transitioned from vegetative to reproductive growth (Brownsey et al. 2017). Reproductive growth can be identified by the onset of the seed head and awns (Brownsey et al. 2017). Mowing or defoliating medusahead to 3 to 6 cm in height during the transition from vegetative to reproductive growth was found to reduce seed production between 80 to 100% (Brownsey et al. 2017).

Tilling or disking the soil is a common strategy in agricultural fields before planting beds of crops (DiTomaso et al. 2007). In natural areas, tilling or disking can be useful to prepare a site for a prescribed burn or the application of herbicides by removing aboveground biomass and breaking up the thatch layer (Schantz et al. 2019). It has also been shown that the removal of thatch with tilling before the application of the herbicide imazapic results in greater control of medusahead (Kyser et al. 2007).

4.4.2 Disadvantages

Mechanical control can cause disturbance to the soil community and increase the spread of certain plants (DiTomaso et al. 2010). For example, plants with underground stems (i.e., rhizomatous grasses) can survive a mechanical control treatment because these plants can grow shoots from broken pieces of stem (Masters and Sheley 2001). The increased amount of bare ground from mechanical control allows additional light penetration to the soil surface and opens a niche that other plants can occupy (DiTomaso et al. 2010). If the mechanical control efforts occur too early in the growing season when the soils still have moisture, there is a greater potential for invasive plants to spread and increase their growth rate as they occupy the open niche (DiTomaso et al. 2007).

Mechanical control is not a selective management strategy and costs can be high over a large area (Masters and Sheley 2001). Mowing and tilling removes all aboveground biomass regardless of the invasive status of a plant. Hand removal of invasive plants can minimize the

impacts on non-target plants (Flory and Clay 2009). However, non-target plants are at risk of being misidentified as an undesirable plant or as unintentional collateral damage. Additionally, hand removal is an extremely labor-intensive removal strategy (Kyser et al. 2014). Organizations that choose to utilize hand removal of invasive plants often rely on volunteer labor (DiTomaso et al. 2007).

4.4.1 Impacts on Plant Community

The small number of studies made it difficult to determine any trends of impacts on non-target plants. Additionally, even though both studies measured the percent cover, the results were presented as transitions between vegetative states. Each vegetative state represents a dominant species or community of species. Berleman et al. (2016) identified the following four vegetative states: medusahead, wild oats, needlegrass mix (i.e., mostly native perennial grasses), and filaree. Similarly, Stein et al. (2016) identified four vegetative states: medusahead, annual exotic forage grasses, native perennial bunchgrasses, and forbs.

Vegetative transitions in a study utilizing both seed-limiting (i.e., mowing) and burning treatments found that seed-limited plots had similar vegetative transitions to the control plots (Table 3) (Berleman et al. 2016). However, plots that were both seed-limited and burned tended to transition to filaree (Berleman et al. 2016). A study using a gradient of mowing intensities found that mowing intensity had no impact on the probability of transitioning between medusahead, native perennial bunchgrasses, or annual forage grasses (Stein et al. 2016). Higher mowing intensities did increase the probability that a vegetative state shifts towards a forb dominated state (Stein et al. 2016). Interestingly, in the non-mowed control plots, native perennial bunchgrass was the only vegetative state that did not transition to another vegetative state (Stein et al. 2016).

Table 3: Transitions between dominant vegetation types from pre-treatment to post-treatment (i.e., year one to year two).

Transition (year one to year two)	Treatment	Number of plots
Medusahead to Needlegrass mix	Seed-limited/not burned	2
Medusahead to Filaree	Seed-limited/not burned	1
Filaree to Medusahead	Seed-limited/not burned	1
Filaree to Wild oat	Seed-limited/not burned	2
Medusahead to Needlegrass mix	Control	2
Medusahead to Filaree	Control	1
Filaree to Medusahead	Control	2
Filaree to Wild oat	Control	1
Medusahead to Filaree	Burned/not seed-limited	7
Needlegrass mix to Filaree	Burned/not seed-limited	1
Filaree to Wild oat	Burned/not seed-limited	1
Medusahead to Filaree	Burned & seed-limited	5
Needlegrass mix to Filaree	Burned & seed-limited	2
Filaree to Wild oat	Burned & seed-limited	1

Adapted from Berleman et al. 2016

Berleman et al. (2016) also measured seed rain or the number of seeds dispersing to the ground. Plots that were mowed, but not burned, had the lowest amount of seed rain (i.e., decreased seed production and dispersal) (Berleman et al. 2016). However, this difference was not significantly different from the control plots (Berleman et al. 2016). Burning and mowing resulted in a seed rain that was slightly higher than the control plots and burning alone had the highest seed rain (i.e., the highest rate of seed dispersal) (Berleman et al. 2016).

5. Discussion

The non-target impacts of managing medusahead are dependent upon many factors. A specific management goal will dictate the timing and management strategy used to control medusahead (Table 4). The studies in this analysis primarily fall under the management goals of reducing seed production or removing thatch. Land managers can also implement strategies to prevent the

spread and reinvasion of medusahead and improve the quality of grasslands that they manage. Additionally, targeting medusahead during the two- to four-week period where it matures later than most other plants will likely result in a reduction of non-target impacts. During this period, most plants are heading towards the end of their life cycle or into dormancy and have already set seed (i.e., reproduced) for the next year.

Table 4: Overview of medusahead management goals and appropriate control strategies

Management Goal	Management Strategy	Considerations	Timing
Reduce seed production	Herbicide, pre-emergent	Potential for dispersal off-site Greater effectiveness with the removal of thatch Greater non-target impacts	Mid to late spring
	Herbicide, post-emergent	Potential for dispersal off-site	Fall
	Mechanical control	Limited by topography Less feasible for large infestations	Mid to late spring
	Prescribed burn	Limited by permits, staffing, and appropriate burn conditions Potential for fire escapes	Early summer
	Grazing	High density, short-duration grazing is optimal At risk of overgrazing or grazing of desirable plants	Mid-spring
Remove thatch	Mechanical removal	Limited by topography Less feasible for large infestations	Mid-spring to early summer
	Prescribed burn	Limited by permits, staffing, and appropriate burn conditions Potential for fire escapes	Late spring to early summer
Prevent reinvasion	Monitoring	Monitor high traffic areas for potential dispersal (i.e., roads, animal paths)	Any time
	Seed hygiene / reduce dispersal	Monitor staff, equipment, animals, and imported goods for medusahead seed heads	Late spring to early summer
Improve grassland	Grazing	Rotational grazing with a high-density stocking over a short period is optimal	Spring
	Revegetation	Success is improved through monitoring and periodic control of medusahead Seedling survival increases if irrigation is provided for one to two years post-planting	Fall or late winter to early spring

Adapted from Kyser et al. 2014

Herbicides were found to have mixed non-target impacts that were influenced by the specific herbicide being applied and the timing of herbicide application (i.e., pre-emergent versus post-emergent). Aminopyralid tends to target forbs and is best suited for sites that have a mixture of medusahead and invasive forbs, such as yellow starthistle. The observed increase in annual grass cover following aminopyralid application is likely due to a release in competitive pressure from medusahead and other dominant forbs (Kyser et al. 2012). Compared to post-emergent application, pre-emergent application of aminopyralid resulted in increased medusahead cover and decreased annual grass cover (Rinella et al. 2018). Conversely, imazapic tends to target grasses, not forbs (Kyser et al. 2007). With imazapic application, there is also the potential for an increase in forb cover due to a reduction in competition from grasses. Rimsulfuron degrades quicker in warmer climates which causes the herbicide to become less effective (Kyser et al. 2012). As observed in this analysis, rimsulfuron had minimal impacts on the percent cover of non-target plants.

Grazing decreased the percent cover of annual grasses and had mixed impacts on the percent cover of native plants. Additionally, grazing at high stocking-densities over a short-time period caused a greater decrease in medusahead cover and resulted in an increase in percent cover of forbs and some native species. The effectiveness of a shorter grazing duration implies that the timing of grazing is an important factor for managing medusahead. Grazing over a longer period at lower densities allows a greater percentage of medusahead to develop into its less palatable growth stages. Grazing at higher stocking densities causes grazers to become less selective with their foraging and forces greater consumption of less palatable plants (Kyser et al. 2014). However, it is difficult for ranchers to achieve the high stocking-densities that were used in the synthesis studies (Davy et al. 2015). One strategy to overcome this barrier would be the implementation of rotational grazing which moves grazers from pasture to pasture and allows for greater control over a grazing area (Roche et al. 2015). Rotational grazing can be labor intensive but allows for higher densities of grazers necessary to target invasive plants while reducing the risk of overgrazing (Roche et al. 2015).

In this analysis, the only grazers were sheep and cattle. Based on the increase in forb cover in grazing studies it can be inferred that cattle and sheep have a preference to forage on grasses. No published studies were found on the effectiveness of goats on the control of medusahead. Goats tend to forage on a greater number of plant species than sheep and cattle and are effective at reducing fuel loads (Lovreglio et al. 2014). Goats have been shown to reduce yellow starthistle cover if grazing occurs during the plant's earliest growth stages (Goehring et al. 2010). However, goats tended to avoid annual grasses in favor of yellow starthistle and other forbs (Goehring et al. 2010). Future research would benefit from using goats in medusahead infested grasslands to determine if goat grazing can be used as a potential management strategy.

Grazing, prescribed burns, and mechanical control all tended to increase the percent cover of non-native forbs. It has been observed that forbs tend to colonize the open space that is left from the removal of annual grasses and thatch (Stahlheber and D'Antonio 2013, Reiner and Craig 2011). In this analysis, one study examining the impact of mowing on vegetative states initially did not account for forbs (Stein et al. 2016). At the beginning of their study they used the classifications of medusahead, annual forage grasses, and native perennial grasses. After the first year of their study, they had to add a category for forbs due to the tendency of forbs to become a dominant vegetative state following mowing treatments. I suspect that the dominance of forbs following the removal of medusahead is partly dependent upon the baseline species composition, both aboveground and in the seed bank. It would be expected that early colonizer species, such as some forbs, already present within the landscape would be the first species to occupy available bare ground.

5.1 Integrated Pest Management

Integrated pest management (IPM) is a science-based approach to managing invasive species that uses a combination of chemical, cultural, mechanical, and biological tools (Masters and Sheley 2001). Many definitions of IPM include an emphasis on shifting away from relying on chemical tools (Masters and Sheley 2001). However, shifting away from herbicide usage in natural areas is not a common approach (Masters and Sheley 2001). The combination of tools used to manage medusahead will depend upon the site location, species composition of the site, management goal, and resources available to the land manager (DiTomaso et al. 2007). Often a single tool is

not effective or sustainable over a long duration to achieve complete control of medusahead. For example, repeated controlled burns are more effective at controlling certain invasive plants, but burns over consecutive years are often prohibited (DiTomaso et al. 2007).

A long-term medusahead management plan will require the integration of complementary management tools to enhance the quality of valley grasslands over time (Figure 14) (DiTomaso et al. 2010). In a medusahead infested site, this may involve the removal of thatch through burning or mowing before a pre-emergent herbicide application (Kyser et al. 2007). Ideally, this site would then be revegetated with native species in the fall to further prevent reinvasion of medusahead. Increasing the quality of valley grasslands through an IPM framework would lead to the enhancement and restoration of ecosystem functions.

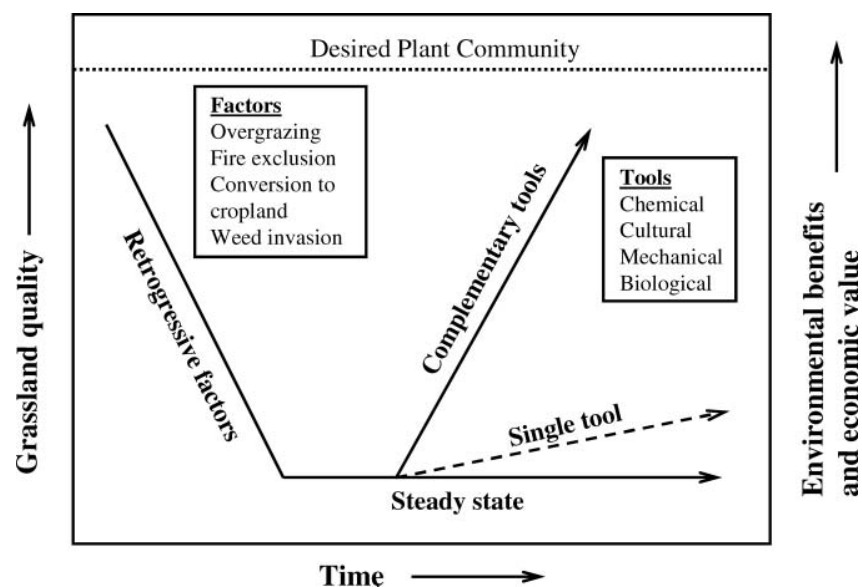


Figure 14: Conceptual framework for integrated pest management in a grassland ecosystem. Retrogressive factors increase open niches available for invasions. The recovery of a grassland is slower with the use of a single tool versus the complementary use of multiple tools (DiTomaso et al. 2010).

5.2 The Knowing-Doing Gap

In invasion biology there is a lack of information flow between land managers and researchers. The barrier of communication between land managers and researchers is often referred to as the knowing-doing gap, or less commonly as the research-implementation gap (Matzek et al. 2014, Li et al. 2020). Land managers tend to rely more on their own and their colleagues' experience, not scientific research (Matzek et al. 2014). Conversely, researchers often do not know what

information land managers need about invasion biology and management (Matzek et al. 2014). A survey of 20 journals focusing on invasion biology found that approximately two thirds of basic research papers (e.g., ecological interactions) provided minimal or no management recommendations (Figure 15A) (Matzek et al. 2015). Around 80% of applied research papers (e.g., controlling invasive species) did not mention the cost of treatments used to control invasive species (Figure 15B) (Matzek et al. 2015). Similarly, all studies within this analysis can be classified as applied research and zero studies reported the cost of treatment. Land managers are not able to make an informed decision on whether to apply current research to their land without information on the practical application of research and the cost to implement control measures.

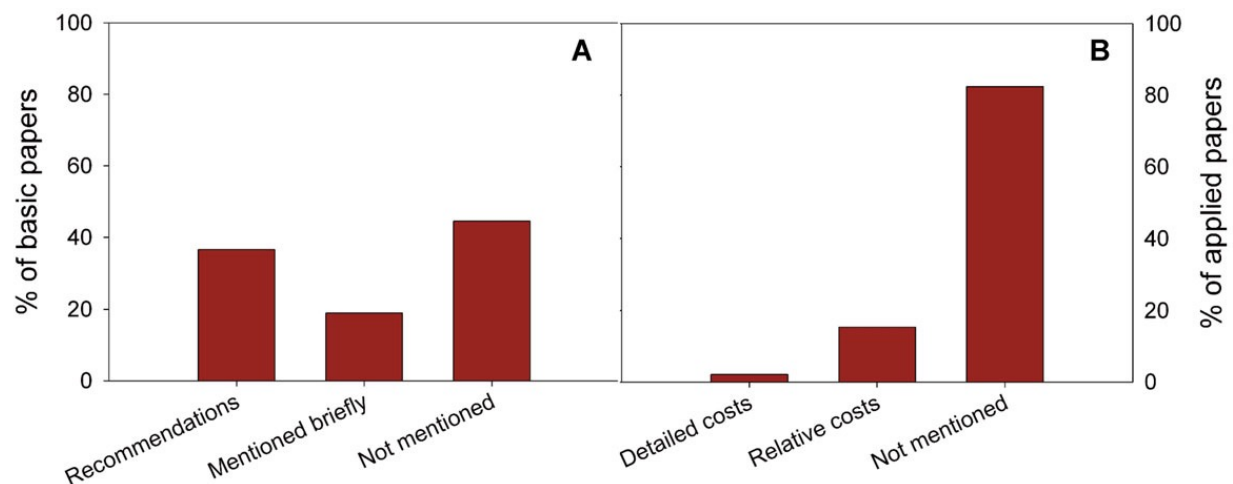


Figure 15: Comparison of usability of scientific literature by land managers. For basic research (A), usability was measured as the level of management recommendations provided by researchers. For applied research (B), usability was measured as the level of information provided on the cost of treatments (Matzek et al. 2015).

To reduce the knowing-doing gap it is imperative to increase communication and build a working relationship between land managers and researchers. Many land managers hold advanced degrees and can navigate the scientific literature (Matzek et al. 2014). However, land managers cite a lack of time (66% of respondents) and a lack of accessibility due to paywalls (50% of respondents) as the main factors preventing them from utilizing research (Matzek et al. 2014). Additionally, many land managers are already collecting informal data on their invasive plant management efforts (Li et al. 2020). While this data is often not adequate for scientific analysis (Matzek et al. 2014), there are opportunities for collaboration between land managers

and researchers. Researchers can create an experimental design and analyze data and have land managers help collect the data (Matzek et al. 2015). Land managers would gain valuable information about managing their land while researchers would gain insights from practitioners. Another strategy to increase information flow between land managers and researchers is the development of training materials or workshops (Gornish and Roche 2018). Training materials could involve the synthesis of current scientific literature or hands-on demonstrations of emerging management strategies. Workshops with multiple stakeholders would also be an effective avenue to provide resources for land managers and update researchers on the biggest priorities for applied research (Gornish and Roche 2018).

5.3 Revegetation Implementation

Revegetating or restoring a site after removing medusahead is not a common practice (James et al. 2015). Of the 14 studies identified in this analysis, zero tested the effectiveness of seeding desirable species after reducing or removing medusahead. However, a couple of studies discussed the importance of revegetating a site after removing medusahead (e.g., Kyser et al. 2007, Kyser et al. 2012). Additionally, it is assumed that the control of medusahead is temporary unless the seed bank at a site has native species (James et al. 2015). While seed banks were not specifically analyzed, study sites that reported the presence of native species found these plants had a low percent cover (e.g., Betts 2003, James et al. 2017). It can be assumed these sites would not shift towards a native species dominated composition without revegetation due to a low abundance of native species in the seed bank.

In practice, not all valley grassland sites can be revegetated for financial and logistical reasons. Sites that have some native plant species should be prioritized for revegetation over sites that are dominated by medusahead monocultures or by several species of invasive plants (Davies and Sheley 2011). If a site is used primarily for grazing, then the determining factor for revegetation may be the presence of desirable forage plants (Davies and Sheley 2011). In these revegetated sites, the goal should not be to permanently eliminate medusahead and other invasive plants. The goal is to create a grassland community with a high functional diversity to increase the overall resistance to reinvasion by medusahead (Lulow 2006). One method used to establish a plant community with high biodiversity is active restoration (Rayburn et al. 2016). Active restoration

involves removal of non-native species, site preparation, seeding or nursery plug planting, and post-planting management (Rayburn et al. 2016). Most valley grassland sites are degraded to the point where active restoration is necessary (Rayburn et al. 2016). Medusahead management strategies that increase the amount of bare ground are ideal for the preparation of restoration plantings (e.g., burning) (Kyser et al. 2014). Continued monitoring and control of medusahead at these restored sites will be necessary to ensure the plant community is progressing towards a predominately native species plant composition (Davies and Sheley 2011).

While revegetating with native plants may be a long-term restoration goal, little is known about the most appropriate native species to use in grassland restoration projects. The most common native perennial grass in many restoration projects is purple needlegrass (Lulow et al. 2007). The life-history traits of purple needlegrass are similar to non-native annual grasses, which make purple needlegrass a decent competitor for resources with these non-native species (Lulow et al. 2007). However, no studies have been published to determine whether stands of purple needlegrass are capable of resisting medusahead invasion. Promising research has found that native annual ryegrasses (e.g., *Elymus glaucus* and *E. triticoides*) are effective at resisting, but not eliminating, medusahead and barbed goatgrass (*Aegilops triuncialis*) from experimental plots (Eviner and Malmstrom 2018). The total percent cover of these two invasive plants averaged around 20% (Eviner and Malmstrom 2018). However, by the end of the six-year study the native ryegrasses were outcompeted and overtaken by non-native grasses (e.g., wild oats and soft brome) (Eviner and Malmstrom 2018).

5.4 Climate Change and Drought

Climate change is predicted to increase global temperatures and the frequency of extreme weather events, such as drought (Diffenbaugh et al. 2015). Valley grasslands are expected to shift in their distribution and experience increasing conversion pressure from urbanization and agriculture (Byrd et al. 2015). Of interest to grassland ecologists is how community processes and individual species will respond to severe drought conditions caused by climate change (e.g., Miller et al. 2019, Harrison et al. 2018, LaForgia et al. 2018). Long-term observational studies in two valley grassland sites have shown that drier winters result in a long-term decrease in species richness (Miller et al. 2019, Harrison et al. 2018). The decrease in species richness was attributed

to a decrease in native forb cover (Miller et al. 2019) and native annual grass cover (Harrison et al. 2018). Conversely, another study looking at seed banks during drought conditions found that non-native grass seeds decreased in abundance over a two-year period while native forb seeds increased in abundance (LaForgia et al. 2018). The shift in seed bank composition corresponded with a decrease in aboveground cover for non-native grasses and an increase in native forb cover (LaForgia et al. 2018). Similarly, one of the studies in this analysis saw a decrease in medusahead cover and an increase in filaree cover in their control plots and proposed that the drought conditions during their experiment contributed to the change in percent cover (Davy and Dykier 2017). Other findings in this analysis were likely influenced by climate and are subject to change in the future due to increasing variability in precipitation patterns and drought conditions.

The conflicting evidence of how valley grasslands will respond to changes in climatic and drought patterns will make managing invasive plants more difficult. Land managers have observed that drought conditions reduce the effectiveness of treatment, increase the potential for invasive plants to become more competitive, and complicate the timing of treatments (Li et al. 2020). Management decisions are further complicated by the uncertainty of how severe droughts will become and how invasive plants will respond to climate change. Further insights and concerns from land managers can inform future research projects on the application of invasive plant control in a shifting climate.

5.5 Study Limitations

This study is limited by the current literature available on medusahead management. The studies included in the analysis were short-term, covered small plot sizes, and did not have a diverse location range. All 14 studies occurred for 6 years or less with 5 studies lasting only one year. Longer studies were able to observe shifts in plant communities that were more likely to be influenced by variation in precipitation than non-target impacts of medusahead control efforts (Davy and Dykier 2017, Davy et al. 2015). Additionally, the plot sizes of the treatments were generally small. The only exception to this was grazing studies which were limited by the size of the ranch. One of the largest treatment plots occurred on a 200-hectare prescribed burn (Davy and Dykier 2017). However, another study using a prescribed burn had treatment plots that were 9 m² with the burn treatment occurring in the middle 1 m² of each plot (Berleman et al. 2016).

Small plot sizes are experimental and do not represent the full spectrum of interactions that occur on a landscape scale. In practice, medusahead management occurs on a larger scale. There may be issues of scaling up management strategies if a land manager is relying on information obtained from a small experimental plot. Lastly, there is a bias in study location. Most of the study sites were located in counties in the north (Tehama, n=4) and central (Yuba and Yolo, n=5 for both) regions of the Central Valley. The only study site in the southern region of the Central Valley was in Fresno County (n=1). Furthermore, all five Yuba County study sites were conducted at the Sierra Foothill Research and Extension Center and four of the Yolo County study sites were conducted at Bobcat Ranch. Location bias potentially reduces the variability of data in this study and lessens the effect size of observed patterns.

Another limitation is this study is the lack of interviews from land managers. As previously discussed, land managers have observations and experiences with medusahead management that scientific experiments are not able to address. Land manager interviews would have also been useful to determine if the financial cost of controlling medusahead restricted which management strategies were utilized for a site. Outreach to land managers was attempted for this study and two responses were obtained. However, one response was from a land manager located outside of the valley grassland range and another response was from an ecologist who primarily manages barbed goatgrass.

6. Conclusion and Management Recommendations

Medusahead management has been shown to cause non-target impacts to valley grassland plant communities. The impacts on non-target plants are generally short-lived and depend upon the timing and management strategy used. The most prevalent impact on non-target plants is a shift in the composition of percent cover from medusahead to either a forb or non-native grass dominated community. Differences in species composition and percent cover between control and treatment plots typically disappear between one to three years after the control treatment. Of the available management strategies for medusahead, herbicides should be used for spot treatments or early infestations of medusahead. Prescribed burns and grazing are both effective at reducing the amount of medusahead thatch. The optimal timing for prescribed burns is when medusahead has started to produce seed heads but seeds have not yet been released, typically in

late spring or early summer. Targeted grazing is best when used with cattle or sheep in the spring before medusahead has produced seeds. Similar to targeted grazing, mowing is most effective when implemented before medusahead has produced seeds.

Most studies in this analysis only performed a one-time control treatment on medusahead that resulted in a short-lived decrease of medusahead cover. Medusahead cover typically increased back to pre-treatment levels one to two years post-treatment. The long-term control of medusahead will require multiple, complementary management strategies that are applied over time. The most crucial aspects of restricting medusahead to its current range involve researching restoration methods, providing consistent funding for regional weed management efforts, and the continued monitoring of medusahead.

6.1 Restoration Research

Research is lacking on the best restoration strategies to employ in valley grasslands to create resilient plant communities. This analysis can be used as a starting point to determine the best strategy to remove medusahead in preparation for revegetation activities. Due to the limited budget of many land management agencies, restoration research should focus on the practicality of utilizing natural regeneration or similar low-cost methods. For example, how practical would it be to grow native species on-site and collect their seeds for future revegetation projects? Additional research should focus on which native plant species are best suited to prevent reinvasion of medusahead. Studies on competition between medusahead and native species can occur through academic studies, student projects, or even by land managers. Another research area could consider whether applying successional ecology theory to restoration planting results in greater resistance to medusahead and an increase in species diversity. For example, would planting a quick-growing, annual grass species prevent medusahead invasion long enough for slower growing, perennial species to establish?

6.2 Funding for Weed Management Areas

Weed Management Areas (WMA) are partnerships formed between land management organizations, government agencies, and other stakeholders to coordinate invasive plant management within a defined area (Ervin and Frisvold 2016). WMA are a powerful tool for

implementing regional invasive plant management plans. WMA can also facilitate the sharing of information and resources among regional stakeholders. However, funding of WMA in California is dependent upon the state budget and has not been consistent over the years (Funk et al. 2014). Additionally, state funding for the program was cut in 2011 and many WMA activities within California came to a halt (Funk et al. 2014). Consistent funding, even at lower monetary levels, has demonstrated greater effectiveness for controlling invasive plants (Funk et al. 2014). The lack of funding for WMA in California has meant that land managers experienced an approximately 50% decline in their annual weed management budgets (Funk et al. 2014). To ensure the future success of controlling medusahead within California, consistent state funding and alternate funding structures should be found for WMA.

6.3 Monitoring and Adaptive Management

The long-term management of medusahead will require the use of monitoring and adaptive management. Monitoring a site after medusahead control will inform a land manager if a specific management strategy is successful and when follow-up treatments are necessary. Monitoring can also identify new infestations of medusahead before they become established. Monitoring should occur before and after treatment efforts and on an annual or bi-annual basis thereafter. Simple monitoring can comprise of quick site visits, ocular scans for shifts in vegetation patterns, and photographs. Photographs are particularly useful to compare the same site year after year. More comprehensive monitoring would add in vegetation sampling of either a permanent plot or transect. Vegetation sampling should consist of species composition and percent cover. With proper training, the collection of monitoring data can often be performed by volunteers and interns. The analysis of monitoring data would inform future management actions to be taken by a land manager.

Adaptive management provides a framework for updating management goals and strategies with the input of data and site conditions through monitoring (Kimball and Lulow 2019). Adaptive management is a repetitive process that typically follows the steps of plan, do, monitor, learn, and adjust (Kimball and Lulow 2019). A simplified example of adaptive management is the early detection of medusahead infestations. Early prevention and eradication are known to be the most cost-effective invasive plant management strategy (Funk et al. 2014). However, early prevention

relies on regular monitoring of a site to identify any new infestations of medusahead. If monitoring does detect a new infestation, a land manager can choose to redistribute their resources from controlling current medusahead populations to the eradication of the new medusahead population. The manager would set a goal (e.g., reduce medusahead cover by X%) and choose a management strategy to control the new medusahead population. Further monitoring would inform the land manager if the treatment was successful, if treatment needed to be repeated, or if an entirely new management strategy needed to be utilized.

References

- Alavanja, M. C. R., J. A. Hoppin, and F. Kamel. 2004. Health effects of chronic pesticide exposure: cancer and neurotoxicity. *Annual review of public health* **25**:155-197.
- Alba, C., H. Skálová, K. F. McGregor, C. D'Antonio, P. Pyšek, and J. Morgan. 2015. Native and exotic plant species respond differently to wildfire and prescribed fire as revealed by meta-analysis. *Journal of Vegetation Science* **26**:102-113.
- Anderson, M. K. 2019. The use of fire by Native Americans in California. Pages 381-398 *In* Van Wagendonk, J. W., N. G. Sugihara, S. L. Stephens, A. E. Thode, K. E. Shaffer, and J. A. Fites-Kaufman, editors. *Fire in California's ecosystems*, University of California Press, Berkeley, California.
- Appleby, A. P. 2005. A history of weed control in the United States and Canada—a sequel. *Weed Science* **53**:762-768.
- Atwill, E. R., K. W. Tate, das Gracas Cabral Pereira, Maria, J. Bartolome, and G. Nader. 2006. Efficacy of natural grassland buffers for removal of *Cryptosporidium parvum* in rangeland runoff. *Journal of Food Protection* **69**:177-184.
- Baldwin, B. G., D. H. Goldman, D. J. Keil, R. Patterson, T. J. Rosatti, and D. Goldman. 2012. *The Jepson manual*. University of California Press, Berkeley, California.
- Bartolome, J. W., W. J. Barry, T. Griggs, and P. Hopkinson. 2007. Valley grassland. Pages 367-393 *In* Barbour, M., A. A. Schoenherr, and T. Keeler-Wolf, editors. *Terrestrial vegetation of California*, 3rd edition, University of California Press, Berkeley, CA.
- Berleman, S. A., K. N. Suding, D. L. Fry, J. W. Bartolome, and S. L. Stephens. 2016. Prescribed fire effects on population dynamics of an annual grassland. *Rangeland Ecology & Management* **69**:423-429.
- Betts, A. D. K. 2003. Ecology and control of goatgrass (*Aegilops triuncialis*) and medusahead (*Taeniatherum caput-medusae*) in California annual grasslands. Berkeley, California: University of California, Berkeley.
- Bovey, R. W., D. Le Tourneau, and L. C. Erickson. 1961. The chemical composition of medusahead and downy brome. *Weeds* **9**:307-311.
- Brooks, M. L., C. M. D'Antonio, D. M. Richardson, J. B. Grace, J. E. Keeley, J. M. DiTomaso, R. J. Hobbs, M. Pellant, and D. Pyke. 2004. Effects of invasive alien plants on fire regimes. *BioScience* **54**:677-688.
- Brownsey, P., J. J. James, S. J. Barry, T. A. Becchetti, J. S. Davy, M. P. Doran, L. C. Forero, J. M. Harper, R. E. Larsen, S. R. Larson-Praplan, J. Zhang, and E. A. Laca. 2017. Using Phenology to Optimize Timing of Mowing and Grazing Treatments for Medusahead (*Taeniatherum caput-medusae*). *Rangeland Ecology & Management* **70**:210-218.

- Burcham, L. T. 1961. Cattle and range forage in California: 1770-1880. *Agricultural History* **35**:140-149.
- Byrd, K. B., L. E. Flint, P. Alvarez, C. F. Casey, B. M. Sleeter, C. E. Soulard, A. L. Flint, and T. L. Sohl. 2015. Integrated climate and land use change scenarios for California rangeland ecosystem services: wildlife habitat, soil carbon, and water supply. *Landscape Ecology* **30**:729-750.
- Byrnes, R. C., D. J. Eastburn, K. W. Tate, and L. M. Roche. 2018. A global meta-analysis of grazing impacts on soil health indicators. *Journal of Environmental Quality* **47**:758-765.
- Calflora: Information on California plants for education, research and conservation, with data contributed by public and private institutions and individuals, including the Consortium of California Herbaria. 2020. *Elymus caput-medusae* L. Berkeley, California. Available: <https://www.calflora.org/cgi-bin/countytaxon.cgi?where-calrecnum=11634> (Accessed: February 15, 2020).
- California Invasive Plant Council. 2017. California invasive plant inventory: plant assessment form. Berkeley, California. Available: <https://www.cal-ipc.org/plants/paf/elymus-caput-medusae-plant-assessment-form/> (Accessed: February 20, 2020)
- Chaplin-Kramer, R., K. Tuxen-Bettman, and C. Kremen. 2011. Value of wildland habitat for supplying pollination services to Californian agriculture. *Rangelands* **33**:33-41.
- Chuong, J., J. Huxley, E. N. Spotswood, L. Nichols, P. Mariotte, and K. N. Suding. 2016. Cattle as dispersal vectors of invasive and introduced plants in a California annual grassland. *Rangeland Ecology & Management* **69**:52-58.
- Clinton, W. J. 1999. Executive order 13112 - invasive species. Weekly Compilation of Presidential Documents **64**:6183-6186.
- Coleman, H., and J. Levine. 2007. Mechanisms underlying the impacts of exotic annual grasses in a coastal California meadow. *Biological Invasions* **9**:65-71.
- Crone, E. E., M. Marler, and D. E. Pearson. 2009. Non-target effects of broadleaf herbicide on a native perennial forb: a demographic framework for assessing and minimizing impacts. *Journal of Applied Ecology* **46**:673-682.
- Crowner, A. W., and G. W. Barrett. 1979. Effects of fire on the small mammal component of an experimental grassland community. *Journal of Mammalogy* **60**:803-813.
- Dahlgren, R. A., K. W. Tate, D. J. Lewis, E. R. Atwill, J. M. Harper, and B. H. Allen-Diaz. 2001. Watershed research examines rangeland management effects on water quality. *California Agriculture* **55**:64-71.
- D'Antonio, C. M., C. Malmstrom, S. A. Reynolds, and J. Gerlach. 2007. Ecology of invasive non-native species in California grassland. Pages 67-83 *In* Stromberg, M. R., J. D. Corbin,

- and C. M. D'Antonio, editors. California grasslands: Ecology and management, University of California Press, Berkeley, California.
- Davies, K. W., and R. L. Sheley. 2011. Promoting native vegetation and diversity in exotic annual grass infestations. *Restoration Ecology* **19**:159-165.
- Davies, K. W. 2011. Plant community diversity and native plant abundance decline with increasing abundance of an exotic annual grass. *Oecologia* **167**:481-491.
- Davies, K. W. 2008. Medusahead dispersal and establishment in sagebrush steppe plant communities. *Rangeland Ecology & Management* **61**:110-115.
- Davies, K. W., and T. J. Svejcar. 2008. Comparison of medusahead-invaded and noninvaded Wyoming big sagebrush steppe in Southeastern Oregon. *Rangeland Ecology & Management* **61**:623-629.
- Davies, K. W., A. M. Nafus, and M. D. Madsen. 2013. Medusahead invasion along unimproved roads, animal trails, and random transects. *Western North American Naturalist* **73**:54-59.
- Davy, J. S., and K. Dykier. 2017. Longevity of a controlled burn's impacts on species composition and biomass in Northern California annual rangeland during drought. *Rangeland Ecology & Management* **70**:755-758.
- Davy, J. S., L. M. Roche, A. V. Robertson, D. E. Nay, and K. W. Tate. 2015. Introducing cattle grazing to a noxious weed-dominated rangeland shifts plant communities. *California Agriculture* **69**:230-236.
- Defossé, G. E., and R. Robberecht. 1996. Effects of competition on the postfire recovery of 2 bunchgrass species. *Journal of Range Management* **49**:137-142.
- Diffenbaugh, N. S., D. L. Swain, and D. Touma. 2015. Anthropogenic warming has increased drought risk in California. *Proceedings of the National Academy of Sciences of the United States of America* **112**:3931-3936.
- DiTomaso, J. M. 2000. Invasive weeds in rangelands: species, impacts, and management. *Weed Science* **48**:255-265.
- DiTomaso, J. M., and B. S. Smith. 2012. Linking ecological principles to tools and strategies in an EBIPM program. *Rangelands* **34**:30-34.
- DiTomaso, J. M., R. A. Masters, and V. F. Peterson. 2010. Rangeland invasive plant management. *Rangelands* **32**:43-47.
- DiTomaso, J. M., S. F. Enloe, and M. J. Pitcairn. 2007. Exotic plant management in California's annual grasslands. Pages 281-296 *In* Stromberg, M. R., J. D. Corbin, and C. M. D'Antonio, editors. California grasslands: ecology and management, University of California Press, Berkeley, California.

- DiTomaso, J. M., G. B. Kyser, and M. S. Hastings. 1999. Prescribed burning for control of yellow starthistle (*Centaurea solstitialis*) and enhanced native plant diversity. *Weed Science* **47**:233-242.
- DiTomaso, J. M., G. B. Kyser, M. R. George, M. P. Doran, and E. A. Laca. 2008. Control of medusahead (*Taeniatherum caput-medusae*) using timely sheep grazing. *Invasive Plant Science and Management* **1**:241-247.
- DiTomaso, J. M., M. L. Brooks, E. B. Allen, R. Minnich, P. M. Rice, and G. B. Kyser. 2006. Control of invasive weeds with prescribed burning. *Weed Technology* **20**:535-548.
- Duncan, C. A., J. J. Jachetta, M. L. Brown, V. F. Carrithers, J. K. Clark, J. M. DiTomaso, R. G. Lym, K. C. McDaniel, M. J. Renz, and P. M. Rice. 2004. Assessing the economic, environmental, and societal losses from invasive plants on rangeland and wildlands. *Weed Technology* **18**:1411-1416.
- Egan, J. F., E. Bohnenblust, S. Goslee, D. Mortensen, and J. Tooker. 2014. Herbicide drift can affect plant and arthropod communities. *Agriculture, Ecosystems and Environment* **185**:77-87.
- Emmerich, W. E., and R. K. Heitschmidt. 2002. Drought and grazing: II. effects on runoff and water quality. *Journal of Range Management* **55**:229-234.
- Ervin, D. E., and G. B. Frisvold. 2016. Community-based approaches to herbicide-resistant weed management. *Weed Science* **64**:609-626.
- Eviner, V. T. 2016. Grasslands. Pages 449-477 In Mooney, H., and E. Zavaleta, editors. *Ecosystems of California*, University of California Press, Berkeley, California.
- Eviner, V. T., and C. Malmstrom. 2018. California's native perennial grasses provide strong suppression of goatgrass and medusahead. *Grasslands* **28**:3-6.
- Fehmi, J., and J. W. Bartolome. 2003. Impacts of livestock and burning on the spatial patterns of the grass *Nassella pulchra* (Poaceae). *Madroño* **50**:8-14.
- Fleischner, T. L. 1994. Ecological costs of livestock grazing in western North America. *Conservation Biology* **8**:629-644.
- Fletcher, R. A., R. K. Brooks, V. T. Lakoba, G. Sharma, A. R. Heminger, C. C. Dickinson, and J. N. Barney. 2019. Invasive plants negatively impact native, but not exotic, animals. *Global Change Biology* **25**:3694-3705.
- Flory, S. L., and K. Clay. 2009. Invasive plant removal method determines native plant community responses. *Journal of Applied Ecology* **46**:434-442.
- FRAP. 2018. California's forests and rangelands: 2017 assessment. California Department of Forestry and Fire Protection Fire and Resource Assessment Program. Sacramento, California.

- Freemark, K., and C. Boutin. 1995. Impacts of agricultural herbicide use on terrestrial wildlife in temperate landscapes: A review with special reference to North America. *Agriculture, Ecosystems and Environment* **52**:67-91.
- Frost, R. A., and K. L. Launchbaugh. 2003. Prescription grazing for rangeland weed management: A new look at an old tool. *Rangelands* **25**:43-47.
- Funk, J. L., V. Matzek, M. Bernhardt, and D. Johnson. 2014. Broadening the case for invasive species management to include impacts on ecosystem services. *BioScience* **64**:58-63.
- George, M. R., R. S. Knight, P. B. Sands, and M. W. Demment. 1989. Intensive grazing increases beef production. *California Agriculture* **43**:16-19.
- Goebel, C. J., and G. Berry. 1976. Selectivity of range grass seeds by local birds. *Journal of Range Management* **29**:393-395.
- Goehring, B. J., K. L. Launchbaugh, and L. M. Wilson. 2010. Late-season targeted grazing of yellow starthistle (*Centaurea solstitialis*) with goats in Idaho. *Invasive Plant Science and Management* **3**:148-154.
- Gornish, E. S., and L. M. Roche. 2018. The value of cooperative extension for involving society in restoration and conservation. *Restoration Ecology* **26**:1051-1054.
- Hamilton, J. G., J. R. Griffin, and M. R. Stromberg. 2002. Long-term population dynamics of native *Nasella* (Poaceae) bunchgrasses in central California. *Madroño* **49**:274-284.
- Harrison, S. P. 1999. Native and alien species diversity at the local and regional scales in a grazed California grassland. *Oecologia* **121**:99-106.
- Harrison, S. P., M. L. LaForgia, and A. M. Latimer. 2018. Climate-driven diversity change in annual grasslands: drought plus deluge does not equal normal. *Global Change Biology* **24**:1782-1792.
- Harrison, S. P., B. D. Inouye, and H. D. Safford. 2003. Ecological heterogeneity in the effects of grazing and fire on grassland diversity. *Conservation Biology* **17**:837-845.
- Harvey, R. G., and F. J. Mazzotti. 2014. The invasion curve: A tool for understanding invasive species management in south florida. University of Florida Cooperative Extension Service, Institute of Food and Agriculture Sciences, EDIS. Gainesville, FL.
- Hironaka, M. 1961. The relative rate of root development of cheatgrass and medusahead. *J Range Manage* **14**:263-267.
- Jackson, R. D., and J. W. Bartolome. 2002. A state-transition approach to understanding nonequilibrium plant community dynamics in Californian grasslands. *Plant Ecology* **162**:49-65.

- James, J. J., J. S. Davy, M. P. Doran, T. Becchetti, P. Brownsey, and E. A. Laca. 2017. Targeted grazing impacts on invasive and native plant abundance change with grazing duration and stocking density. *Rangeland Ecology & Management* **70**:465-468.
- James, J. J., E. S. Cornish, J. M. DiTomaso, J. S. Davy, M. P. Doran, T. Becchetti, D. Lile, P. Brownsey, and E. A. Laca. 2015. Managing medusahead (*Taeniatherum caput-medusae*) on rangeland: a meta-analysis of control effects and assessment of stakeholder needs. *Rangeland Ecology & Management* **68**:215-223.
- Jantz, P. A., B. F. L. Preusser, J. K. Fujikawa, J. A. Kuhn, C. J. Bersbach, J. L. Gelbard, and F. W. Davis. 2007. Regulatory protection and conservation. Pages 297-318 *In* Stromberg, M. R., J. D. Corbin, and C. M. D'Antonio, editors. California grasslands: ecology and management, University of California Press, Berkeley, California.
- Jensen, M. N. 2001. Can cows and conservation mix? *BioScience* **51**:85-90.
- Kauffman, J. B., and W. C. Krueger. 1984. Livestock impacts on riparian ecosystems and streamside management implications... a review. *Journal of Range Management* **37**:430-438.
- Keeler-Wolf, T., J. M. Evens, A. I. Solomeshch, V. L. Holland, and M. G. Barbour. 2007. Community classification and nomenclature. Pages 21-34 *In* Stromberg, M., J. D. Corbin, and C. M. D'Antonio, editors. California grasslands: ecology and management, University of California Press, Berkeley, California.
- Kimball, S., and M. E. Lulow. 2019. Adaptive management in variable environments. *Plant Ecology* **220**:171-182.
- Kolden, C. A. 2019. We're not doing enough prescribed fire in the western United States to mitigate wildfire risk. *Fire* **30**:1-10.
- Kral, K. C., R. F. Limb, J. P. Harmon, and T. J. Hovick. 2017. Arthropods and fire: previous research shaping future conservation. *Rangeland Ecology & Management* **70**:589-598.
- Krausman, P. R., D. E. Naugle, M. R. Frisina, R. Northrup, V. C. Bleich, W. M. Block, M. C. Wallace, and J. D. Wright. 2009. Livestock grazing, wildlife habitat, and rangeland values. *Rangelands* **31**:15-19.
- Kruse, A. D., and J. L. Piehl. 1986. The impact of prescribed burning on ground-nesting birds. Pages 153-156 *In* Clambey, G. K. and R. H. Pemble, editors. The prairie: past, present and future: Proceedings, 9th North American Prairie Conference, Moorhead, MN, Center for Environmental Studies, Tri-College University, Fargo, North Dakota.
- Kyser, G. B., R. G. Wilson, J. Zhang, and J. M. DiTomaso. 2013. Herbicide-assisted restoration of Great Basin sagebrush steppe infested with medusahead and downy brome. *Rangeland Ecology & Management* **66**:588-596.

- Kyser, G. B., V. F. Peterson, J. S. Davy, and J. M. DiTomaso. 2012. Preemergent control of medusahead on California annual rangelands with aminopyralid. *Rangeland Ecology & Management* **65**:418-425.
- Kyser, G. B., J. DiTomaso, K. Davies, J. S. Davy, and B. Smith. 2014. Medusahead management guide for the western states. Weed Research and Information Center, University of California.
- Kyser, G. B., M. P. Doran, N. K. McDougald, S. B. Orloff, R. N. Vargas, R. G. Wilson, and J. M. DiTomaso. 2008. Site characteristics determine the success of prescribed burning for medusahead (*Taeniatherum caput-medusae*) control. *Invasive Plant Science and Management* **1**:376-384.
- Kyser, G. B., J. M. DiTomaso, M. P. Doran, S. B. Orloff, R. G. Wilson, D. L. Lancaster, D. F. Lile, and M. L. Porath. 2007. Control of medusahead (*Taeniatherum caput-medusae*) and other annual grasses with imazapic. *Weed Technology* **21**:66-75.
- LaForgia, M. L., M. J. Spasojevic, E. J. Case, A. M. Latimer, and S. P. Harrison. 2018. Seed banks of native forbs, but not exotic grasses, increase during extreme drought. *Ecology* **99**:896-903.
- Lampert, A., A. Hastings, E. D. Grosholz, S. L. Jardine, and J. N. Sanchirico. 2014. Optimal approaches for balancing invasive species eradication and endangered species management. *Science* **344**:1028.
- Lavin, M. 2003. *Taeniatherum caput-medusae*. Retrieved from: https://www.flickr.com/photos/plant_diversity/3873644920/
- Lavin, M. 2007. *Taeniatherum caput-medusae*. Retrieved from: https://www.flickr.com/photos/plant_diversity/3873651200
- Li, Y. M., L. M. Roche, and E. S. Gornish. 2020. Bridging the research-implementation gap in weed management on California rangelands. *Rangeland Ecology & Management* In press.
- Liao, C., R. Peng, Y. Luo, X. Zhou, X. Wu, C. Fang, J. Chen, and B. Li. 2008. Altered ecosystem carbon and nitrogen cycles by plant invasion: a meta-analysis. *The New Phytologist* **177**:706-714.
- Longland, W. S. 1994. Seed use by desert granivores. Pages 233–237 In Monsen, S. B., and S. G. Kitchen, editors. Proceedings of the symposium on ecology, management, and restoration of intermountain annual rangelands. USDA Forest Service **INT-GTR-313**
- Lovreglio, R., O. Meddour-Sahar, and V. Leone. 2014. Goat grazing as a wildfire prevention tool: a basic review. *iForest - Biogeosciences and Forestry* **7**:260-268.
- Lulow, M. E. 2006. Invasion by non-native annual grasses: the importance of species biomass, composition, and time among California native grasses of the Central Valley. *Restoration Ecology* **14**:616-626.

- Lulow, M. E., T. P. Young, J. L. Wirka, and J. H. Anderson. 2007. Variation in the initial success of seeded native bunchgrasses in the rangeland foothills of Yolo County, California. *Ecological Restoration* **25**:20-28.
- Mack, R. N., D. Simberloff, W. M. Lonsdale, H. Evans, M. Clout, and F. A. Bazzaz. 2000. Biotic invasions: causes, epidemiology, global consequences, and control. *Ecological Applications* **10**:689-710.
- Marin Municipal Water District. 2019. Draft program environmental impact report for the biodiversity, fire, and fuels integration plan. Corte Madera, California.
- Mariotte, P., E. N. Spotswood, E. C. Farrer, K. N. Suding, and L. Fraser. 2017. Positive litter feedbacks of an introduced species reduce native diversity and promote invasion in Californian grasslands. *Applied Vegetation Science* **20**:28-39.
- Marty, J. 2015. Fire effects on plant biodiversity across multiple sites in California vernal pool grasslands. *Ecological Restoration* **33**:266-273.
- Masters, R. A., and R. L. Sheley. 2001. Principles and practices for managing rangeland invasive plants. *J Range Manage* **54**:502-517.
- Matzek, V., M. Pujalet, and S. Cresci. 2015. What managers want from invasive species research versus what they get. *Conservation Letters* **8**:33-40.
- Matzek, V., J. Covino, J. L. Funk, and M. Saunders. 2014. Closing the knowing-doing gap in invasive plant management: accessibility and interdisciplinarity of scientific research. *Conservation Letters* **7**:208-215.
- Menke, J. W. 1992. Grazing and fire management for native perennial grass restoration in California grasslands. *Fremontia* **20**:22-25.
- Miller, J. E. D., D. Li, M. LaForgia, S. P. Harrison, and H. Jones. 2019. Functional diversity is a passenger but not driver of drought-related plant diversity losses in annual grasslands. *Journal of Ecology* **107**:2033-2039.
- Monaco, T. A., T. M. Osmond, and S. A. Dewey. 2005. Medusahead control with fall- and spring-applied herbicides on northern Utah foothills. *Weed Technology* **19**:653-658.
- Murphy, A. H., and W. C. Lusk. 1961. Timing medusahead burns. *California Agriculture* **15**:6-7.
- Nafus, A. M., and K. W. Davies. 2014. Medusahead ecology and management: California annual grasslands to the Intermountain West. *Invasive Plant Science and Management* **7**:210-221.
- Norgaard, K. M. 2007. The politics of invasive weed management: gender, race, and risk perception in rural California. *Rural Sociology* **72**:450-477.
- Olszyk, D., T. Pflieger, T. Shiroyama, M. Blakeley-Smith, E. H. Lee, and M. Plocher. 2017. Plant reproduction is altered by simulated herbicide drift to constructed plant communities. *Environmental Toxicology and Chemistry* **36**:2799-2813.

- Overton, C., M. Casazza, J. Takekawa, D. Strong, and M. Holyoak. 2014. Tidal and seasonal effects on survival rates of the endangered California clapper rail: does invasive spartina facilitate greater survival in a dynamic environment? *Biological Invasions* **16**:1897-1914.
- Pimentel, D., R. Zuniga, and D. Morrison. 2005. Update on the environmental and economic costs associated with alien-invasive species in the United States. *Ecological Economics* **52**:273-288.
- Pitt, M. D., and H. F. Heady. 1978. Responses of annual vegetation to temperature and rainfall patterns in northern California. *Ecology* **59**:336.
- Pollack, O., and T. Kan. 1998. The use of prescribed fire to control invasive exotic weeds at Jepson Prairie Preserve. Pages 241-249 *In* Witham, C. W., E. T. Bauder, D. Belk, W. R. Ferren Jr., and R. Ornduff, editors. Ecology, Conservation, and Management of Vernal Pool Ecosystems – Proceedings from a 1996 Conference. California Native Plant Society, Sacramento, CA.
- Power, E. F., D. L. Kelly, J. C. Stout, and C. Ghera. 2013. The impacts of traditional and novel herbicide application methods on target plants, non-target plants and production in intensive grasslands. *Weed Research* **53**:131-139.
- Quinn-Davidson, L. N., and J. M. Varner. 2012. Impediments to prescribed fire across agency, landscape and manager: an example from northern California. *International Journal of Wildland Fire* **21**:210.
- Rayburn, A. P., C. Schriefer, A. Zamora, and E. A. Laca. 2016. Seedbank-vegetation relationships in restored and degraded annual California grasslands: implications for restoration. *Ecological Restoration* **34**:277-284.
- Reiner, R., and A. Craig. 2011. Conservation easements in California blue oak woodlands: testing the assumption of livestock grazing as a compatible use. *Natural Areas Journal* **31**:408-413.
- Rinella, M. J., B. D. Maxwell, P. K. Fay, T. Weaver, and R. L. Sheley. 2009. Control effort exacerbates invasive-species problem. *Ecological Applications* **19**:155-162.
- Rinella, M. J., J. S. Davy, G. B. Kyser, F. E. Mashiri, S. E. Bellows, J. J. James, and V. F. Peterson. 2018. Timing aminopyralid to prevent seed production controls medusahead (*Taeniatherum caput-medusae*) and increases forage grasses. *Invasive Plant Science and Management* **11**:61-68.
- Roche, L. M., B. B. Cutts, J. D. Derner, M. N. Lubell, and K. W. Tate. 2015. On-ranch grazing strategies: context for the rotational grazing dilemma. *Rangeland Ecology & Management* **68**:248-256.

- Schantz, M., R. Sheley, and S. Hardegree. 2019. Restoring perennial grasses in medusahead habitat: role of tilling, fire, herbicides, and seeding rate. *Rangeland Ecology & Management* **72**:249-259.
- Schohr, T. K., E. S. Gornish, G. Woodmansee, J. Shaw, K. W. Tate, and L. M. Roche. 2020. Practitioner insights into weed management on California's rangelands and natural areas. *Environmental Management* **65**:212-219.
- Sharp, L. A., M. Hironaka, and E. W. Tlisdale. 1957. Viability of medusa-head (*Elymus caput-medusae* L.) seed collected in Idaho. *J Range Manage* :123-126.
- Shaw, M. R., L. Pendleton, D. R. Cameron, B. Morris, D. Bachelet, K. Klausmeyer, J. MacKenzie, D. R. Conklin, G. N. Bratman, J. Lenihan, E. Haunreiter, C. Daly, and P. R. Roehrdanz. 2011. The impact of climate change on California's ecosystem services. *Climatic Change* **109**:S465-S484.
- Sheley, R. L., and B. S. Smith. 2012. Prioritizing invasive plant management strategies. *Rangelands* **34**:11-14.
- Sheley, R. L., M. F. Carpinelli, and K. J. Reeve Morghan. 2007. Effects of imazapic on target and nontarget vegetation during revegetation. *Weed Technology* **21**:1071-1081.
- Skurski, T. C., B. D. Maxwell, and L. J. Rew. 2013. Ecological tradeoffs in non-native plant management. *Biological Conservation* **159**:292-302.
- Stahlheber, K. A., and C. M. D'Antonio. 2013. Using livestock to manage plant composition: A meta-analysis of grazing in California Mediterranean grasslands. *Biological Conservation* **157**:300-308.
- Stein, C., W. Stanley Harpole, and K. N. Suding. 2016. Transitions and invasion along a grazing gradient in experimental California grasslands. *Ecology* **97**:2319-2330.
- Sweet, S. B., G. B. Kyser, and J. M. DiTomaso. 2008. Susceptibility of exotic annual grass seeds to fire. *Invasive Plant Science and Management* **1**:158-167.
- Tate, K. W., E. R. Atwill, J. W. Bartolome, and G. Nader. 2006. Significant *Escherichia coli* attenuation by vegetative buffers on annual grasslands. *Journal of Environmental Quality* **35**:795-805.
- Tatum, V. L. 2004. Toxicity, transport, and fate of forest herbicides. *Wildlife Society Bulletin* **32**:1042-1048.
- Timmons, F. L. 2005. A history of weed control in the United States and Canada. *Weed Science* **53**:748-761.
- USDI BLM. 1999. Lowden ranch prescribed fire review. USDI Bureau of Land Management. Lewiston, CA.

- Wagner, V., P. M. Antunes, M. Irvine, C. R. Nelson, and J. Firn. 2017. Herbicide usage for invasive non-native plant management in wildland areas of North America. *Journal of Applied Ecology* **54**:198-204.
- Wilcove, D. S., D. Rothstein, J. Dubow, A. Phillips, and E. Losos. 1998. Quantifying threats to imperiled species in the United States. *BioScience* **48**:607-615.
- Young, J. A. 1992. Ecology and management of medusahead (*Taeniatherum caput-medusae* ssp. *asperum* [Simk.] Melderis). *The Great Basin Naturalist* **52**:245-252.
- Zedler, P. H. 2003. Vernal pools and the concept of "isolated wetlands". *Wetlands* **23**:597-607.